APPENDICES A-G, Request for Proposal:
Scientific Literature Review of Forest Management Effects on Riparian Functions in Anadromous Salmonid Fisheries

Introduction

Review of the scientific literature should focus on the interaction between riparian function, stream characteristics, and impacts, both positive and negative, from nearby forest management activities. The life cycle needs of salmonids are well understood and have been documented in the Scientific Review Panel report of 2000 (SRP) and in the Primer found in the Appendices for each Riparian Exchange Function. The Key Questions will focus on riparian function and potential impacts from forest management activities, but the linkage to salmon biology is important. The initial focus of riparian Forest Practice Regulations is to limit or avoid significant impacts to existing riparian function and habitat regardless of specific salmon species needs. Enhancement or restoration of specific water quality or instream elements is a much more site specific analysis.

Each Key Riparian Function and subsequent Key Question is structured following a specific format. Most have "overarching" Key Questions asking whether the scientific understanding of riparian unction is supported by research and then at what distance(s) from the stream channel does the feature contribute to the stream channel. Additional sub questions are included that give a more specific bearing on forest management activities or a specific management buffer protection response. These questions may provide insight into understanding the larger ecological role of the riparian function feature.

In addition to understanding the Key Riparian Function, it is also important to assess natural variability and other more stochastic natural events. Natural Variability from watershed to watershed is typically due to a watersheds unique physical condition (i.e. space) and unique history of natural disturbances (i.e. time). The physical condition of a watershed can vary due to, but not limited to; geology, climate, precipitation patterns and resulting vegetation types. Riparian habitats are typically changed by natural disturbances such as fire, flooding, and windthrow. Stream channels can be changed by disturbances such as landslides, lateral channel erosion, peak flow flooding, and deposition of debris during peak flows. All of these disturbances help create a highly diverse riparian plant communities and complex stream channel habitats (*Gregory et al. 1991*). Accordingly, uncertainty caused by stochastic events requires that riparian habitats and stream channel protection measures be reviewed and assessed on a site-by-site basis as described in the Forest Practice Rules. The natural disturbance history is our attempt to measure time, but in fact the watershed or stream channel is in constant change over time and review of scientific information should reflect this variability.

Appendix A: Biotic and Nutrient Riparian Exchange Function

Primer, Key Questions and Initial List of Literature to be reviewed

Primer
on
Biotic & Nutrient
Riparian Exchanges Related to Forest Management in the Western U.S.

Prepared by the
Technical Advisory Committee
of the
California Board of Forestry and Fire Protection

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Version 1.0

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PRIMER: BIOTIC AND NUTRIENT RIPARIAN EXCHANGE FUNCTION

The riparian vegetation area (zone) along forested streams serves critical biotic and nutrient transfer and exchange functions that directly and indirectly control the survival and growth of juvenile salmonids (e.g. Wilzbach et al. 2005, Jones et al. 2006). Therefore, the timing, magnitude, and qualitative aspects of these biotic and nutrient riparian influences are not only among the very best predictors of overall stream ecosystem health and the condition of the component salmonid populations (e.g. Naiman and Dechamps 1997, Gregory et al. 1991, Meyer et al. 2003, Moore and Richardson 2003), but they also constitute significant potential for management procedures to sustain and/or enhance these salmonid populations (e.g. Bilby and Bisson 1992).

The riparian biotic and nutrient transfers and exchanges are directly or indirectly important to the growth and survival of juvenile salmonids. These can be categorized into: 1) light and nutrients (including dissolved organics), and 2) inputs of particulate organic matter and terrestrial invertebrates (see Figure 1). The general characteristics of the biotic and nutrient exchanges and transfers differ in a predictable way along a west to east gradient. For example, temperature is moderated by coastal climate and has less seasonal effect on in-stream metabolic rates of the resident organisms than in eastern drainages where both daily and seasonal temperature excursions are significantly greater.

Shading by Riparian Vegetation Cover Over, and Transfer of Nutrients into, Streams

Light and nutrients regulate in-stream plant growth, primarily algae. The periphyton assemblage on surfaces in running water constitute the food resource for a group of aquatic invertebrates termed scrapers, after their behavior of scraping loose their attached algal food resource. Light has been shown to be limiting for algal growth in some shaded forest streams even under conditions of very low nutrient concentrations (Gregory 1980, 1983). Limitation of algal growth whether by nitrogen or phosphorous is primarily a function of the parent geology in a watershed (Allan 1995). If light and/or nitrogen and/or phosphorous nutrients become available in significant excess over natural conditions, the algal community can move through a succession from a single cell and small colony community, largely of diatoms and green algae, to a filamentous colony dominated by blue-green (cyanobacteria) and green algae (Stockner and Shortreed 1978, Shortreed and Stockner 1983). The former provides a suitable food resource for scraper invertebrates, the latter does not (e.g. Dudley et al. 1986). Therefore, management actions that shift the periphyton to domination by filamentous forms has a severe negative impact on scrapers, some of which are important prey of juvenile salmonids. Increase of nutrients and light, especially if combined with the deposition of fine sediments, can favor the development of rooted vascular aquatic plants (Clarke 2002). These vascular hydrophytes, including aquatic mosses, if they are present, function primarily as habitat for many invertebrates (e.g. Fisher and Carpenter 1976). That is, they are sites for attachment and concealment, and serve as a food resource for only a very few, and these invertebrates are not commonly consumed by juvenile salmonids (Merritt and Cummins 1996). However, many of the invertebrate taxa that utilize vascular hydrophytes as a habitat are consumed by fish (Svendsen et al. 2004). When filamentous algae and vascular hydrophytes die, they enter the detrital cycle and are consumed by gathering collector invertebrates, many of which are important food organism for juvenile salmonids (Svendsen et al. 2004). A simple and effective bioassay for nitrate and/or phosphate nutrient limitation of algal growth in streams has been developed and well tested (Fairchild and Lowe 1984). Diffusing substrates are used which can be evaluated visually (or by chlorophyll analysis) to determine if a given riparian condition is fostering light and/or nutrient limitation, and, if the latter, which nutrient is most limiting.

Along with nitrogen and phosphorous, dissolved organic matter (DOM) can stimulate the growth of microorganisms that are responsible for the direct decomposition of particulate organic matter (POM) (Ward and Aumen 1986). These microbes also serve as the most important component of the coarse particulate organic matter (CPOM) food source of shredder macroinvertebrates and some of these are prey for juvenile salmonids (Cummins et al. 1989, Svendsen et al. 2004).

Transfer of Riparian Litter and Terrestrial Invertebrates into Streams

Litter derived from riparian vegetation is the dominant base of food chains in forested streams of orders 0 through 3. (Cummins et al.198, Cummins 2002). Up to 90% of the energy flow in such streams is attributable to this litter (Fisher and Likens 1973, Richardson et al. 2006). The processing times (normalized for temperature by expressing it as degree-days) of coarse litter, primarily leaves and needles, is known for a wide range of riparian plant species (Petersen and Cummins 1974, Webster and Benfield 1986, Cummins et al. 1989, Richardson et al. 2004). Riparian litter can be classified according to its processing rate, that is, the turnover time required to convert the material to some other form once it is in the stream. Most hard woods (e.g. alders, vine and big-leaf maples and some shrubs such as salmon berry and elder berry) have short processing times and are referred to as fast (turnover) litter (Petersen and Cummins 1974). By contrast, most conifers (e.g. redwood, Douglas fir) and broad-leaf evergreens (e.g. rhododendron and laurel), oak hardwoods, and willows have long processing times and are termed slow (turnover) litter (Petersen and Cummins 1974). Processing is defined as the sum of leaching of DOM, decomposition by microbes, feeding by shredder invertebrates, and mechanical fragmentation (Cummins et al. 1989). The majority of leaching of soluble organics from wetted litter is rapid with the litter losing 20-40% of it's dry mass in 24 to 72 hours (Petersen and Cummins 1974). This portion of litter processing is non-biological and and fairly independent of temperatures from 5 to 20 ºC (Petersen and Cummins 1974, Dahm 1981). After the initial loss rapid loss of weight due to leaching, small amounts of DOM continue to leach slowly from litter and large woody debris (LWD; Cummins et al. 1983). The riparian terrestrial soil and litter also continuously leach small to moderate amounts of DOM into streams (Allan 1995).

In order for riparian litter to be processed by microbes and shredders it must be retained in place in a given reach for a sufficient period for microbial conditioning and shredder feeding to take place. Small woodland streams have been shown to be quite retentive, providing that sufficient wood debris and other obstructions are present. Once it is wetted, the major portion of the riparian litter introduced into a small stream is retained within the range of 100 meters (Cummins et al. 1989). The percent cover by species of riparian vegetation has been shown to be a good predictor of the percent composition of the litter entrained in a reach of stream. Linked to this, the hatching and major feeding by resident shredder invertebrates is keyed to the timing of the drop and entrainment of the different riparian species (Grubbs and Cummins 1986; Cummins et al. 1989, Richardson 2001)

The end result of litter processing is microbial and invertebrate biomass and fine particulate organic matter (FPOM, <1mm>0.5 µm particle size) (Cuffney et al. 1990). FPOM transported in suspension is the major food of filtering collector invertebrates and, when it settles out on or into the sediments it is the food of gathering collector invertebrates (Merritt and Cummins 1996). These two invertebrate groups contain the most important prey items for juvenile salmonids (Wilzbach et al. 2006).

The aquatic invertebrates that depend upon periphyton, plant litter, and FPOM as their food resources, and constitute important prey for juvenile salmonids in forested streams are tightly coupled to the riparian area, because of the restriction of algal populations by shading and organic matter transfers. The aquatic insects among these can be characterized as having deterministic life cycles that are adapted to stochastic environmental conditions such as flow and temperature regimes and the timing of riparian litter inputs. The general pattern is one in which the most vulnerable life stages are matched to the seasonal periods during which environmental conditions have the highest probability of being favorable (e.g. Fisher et al. 1982). Stream flows suitable to allow eggs and newly hatched nymphs and larvae to maintain their location and the availability of food for feeding nymphs and larva are seasonally timed (Grubbs and Cummins 1996, Richardson 2001). For example, invertebrate shredders lay their eggs in late summer and early fall when stream are at base flow. This timing leads to hatching of larvae and nymphs at the time of abscission of deciduous riparian hardwoods that are in the fast processing category and the food supply of the autumn-winter shredders (Grubbs and Cummins 1996, Cummins et al. 1989). Spring –summer shredder populations rely on litter with longer processing times, such as conifer needles, as their food resource (Cummins, et al.1989, Robinson et al. 2000).

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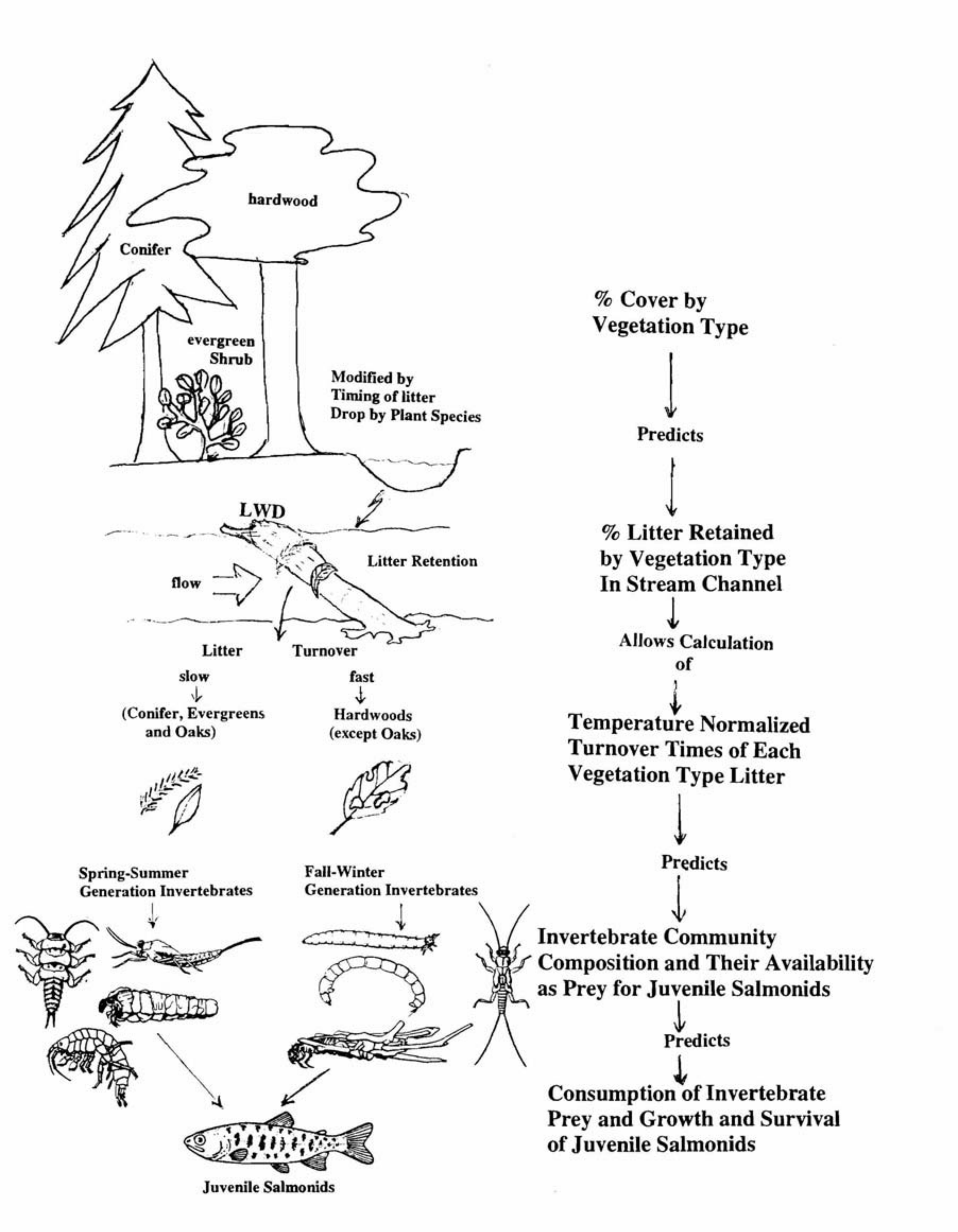
Terrestrial invertebrates also constitute transfers from the riparian area into the stream ecosystem. Included are canopy insects and their frass, annelids, spiders, and ants from the soil and terrestrial litter mat (Nakano and Murakami 2001, Allan et al. 2003). Among the terrestrial invertebrate inputs from the riparian area are the adult (and in some cases pupal) stages of aquatic insects. All of these transfers of terrestrial invertebrates to the stream can serve as important food sources for juvenile salmonids, at least seasonally. Aquatic invertebrates are more abundant in the winter and terrestrial forms are more abundant in the summer in juvenile salmonid diets. (Shigeru and Murakami 2001, Allan et al. 2003).

The activities of the microbes and invertebrate shredders on leaf litter, the resulting FPOM that is generated, and the ensuing effect on invertebrate collectors in the smallest streams is transmitted down stream (e.g. Vannote et al. 1980, Webster et al. 1999, Cummins and Wilzbach 2005, Meyer et al. 2007). Woody debris is also a source of FPOM, although it is released more slowly (Ward and Aumen 1986). These cumulative effects from small headwater streams to larger tributaries constitute an important delivery system to juvenile salmonid populations down stream (e.g. Wipfli and Gregovich 2002, Wipfli and Musselwhite 2004) and constitute a basis for their protection (Cummins and Wilzbach 2005).

Figure 1: Riparian biotic and nutrient transfers and exchanges process relative to growth and survival of juvenile salmonids



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KC 1/23/07

KEY QUESTIONS: BIOTIC AND NUTRIENT RIPARIAN EXCHANGE FUNCTION

The need to resolve uncertainties involving riparian biotic and nutrient transfers and exchanges before the available scientific information is applied to management prescriptions can be captured by one overarching question.

Once objectives have been clearly identified (e.g. faster growth rates or higher densities of juvenile salmonids), what riparian plant species mix, stand age structure, and stem density are optimal for achieving the objectives for a specific species of juvenile salmonid? These are rhetorical questions and objectives that are meant to be established by the Board, not the performing Entity. However, to the extent the performing Entity can identify these objectives in literature reviewed as part of this contract, information on objectives stated in the literature should be disclosed.

This overarching question can be resolved into specific Key Questions given below. Embedded in the answer to these Key Questions must be the following:

A. How does geographic setting modify the answer to the Key Question in hand?

B. How does stream size modify the answer to the Key Question in hand?

C. How does the context for comparison along a gradient from least disturbed to most disturbed modify the answer to the Key Question in hand?

D. How do the forest management practices being examined relate to current California forest practices in the context of the modify the answer to the Key Question in hand? and

E. How do the alterations of the riparian area relate to salmonid habitat quality and salmonid feeding efficiency modify the answer to the Key Question in hand?

Questions Concerning Shading by Riparian Vegetation Cover Over, and Transfer of Nutrients Into the Stream

1. **How can management (manipulation) of the riparian area lead to the establishment and maintenance of algal stream communities most beneficial to juvenile salmonids?**

   a. What riparian stand characteristics are most likely to produce light and nutrient conditions that favor a periphyton cover dominated by diatoms and single-cell or small colony green algae but will avoid (that is, remain below the threshold for) a community shift to filamentous algal forms?

      [Explanation: this is based on the background that a non-filamentous diatom-green algae mix is best at supporting invertebrate scraper populations, which include important food organisms for juvenile salmonids, and that a filamentous-dominated periphyton supports few if any important invertebrate prey of juvenile salmonids.]

Questions Concerning the Vegetative Characteristics of the Riparian Area

2. **How can management (manipulation) of the riparian area lead to rapid processing (turnover) of riparian litter in the stream?**

a. What riparian vegetation stand characteristics are most likely to produce nutrient conditions that favor the development and rapid growth of hyphomycete fungi colonizing leaf/needle litter?

[Explanation: this is based on the background that hyphomycete fungi and associated bacteria that colonize and mineralize riparian-derived litter control the rate of utilization of the litter by shredder invertebrates and, therefore, the rate of FPOM generation. FPOM is an important component of the food of collector invertebrates which include the majority of the aquatic-based prey of juvenile salmonids.]

3. **How can management (manipulation) of the riparian area produce and maintain a mix of litter inputs that favors the components of invertebrate prey organisms to yield higher growth rates and densities of juvenile salmonids?**

a. What riparian vegetation stand characteristics are most likely to produce the best mix of fast (rapid processing rates) and slow (slow processing rates) of litter transferred to the stream?

[Explanation: this is based on the background that fast litter supports populations of invertebrates that feed and grow during the fall and winter and slow litter supports those that feed and grow during the spring and summer. Generation of FPOM from CPOM in all seasons favors year around growth and production of collector invertebrates (which include the majority of the important aquatic-based prey of juvenile salmonids), which in turn favors the growth and survival of juvenile salmonids]

4. **How can management (manipulation) of the riparian area produce and maintain a vegetation mix that favors the availability of terrestrial invertebrates to provide food for juvenile salmonids?**

a. What mix of riparian vegetation is most likely to produce the best populations of terrestrial invertebrates that are an important seasonal food source for juvenile salmonids?

[Explanation: this is based on the background that different species of vegetation have differing amounts of terrestrial invertebrates associated with their foliage, stems, and other plant parts as well as with their terrestrial litter on the forest floor. Juvenile salmonids (growth and survival) are supported directly by terrestrial insects that serve as prey, and indirectly by insect frass that forms a component of FPOM that is food for collector invertebrate populations that are prey for juvenile salmonids.]

Questions Concerning Buffer Width

5. **What riparian buffer width is required to achieve desired conditions of algal growth (question 1), litter turnover (question 2), and invertebrate prey for juvenile salmonids (questions 3 and 4)?**

6. **What valley configurations (e.g. side slopes) and geomorphological characteristics (LWD, sediments, channel structures) set the boundaries for the buffer width required to achieve the objectives in question 5?**

    a. What geomorphic channel and side slope characteristics are important in setting the width of the riparian area (buffer) that?

    [Explanation: this is based on the background that the characteristics of the riparian area vegetation are responsible for transfers that influence the in-stream biology leading to the production of prey for juvenile salmonids.]

Questions Concerning Forest Management Practices and Natural Disturbance

7. **Given a designated riparian buffer width necessary to achieve desired in-steam biological objectives (questions 5 and 6), what have timber operations and management practices in riparian areas have been demonstrated to favor or inhibit these objectives?**

    a. How have selective harvesting and operations at differing distances from stream channel bankfull enhanced or inhibited the development of stream invertebrate communities that favor increased growth and density of juvenile salmonids?

    [Explanation: this is based on the background that species-specific riparian vegetation cover is a good predictor of the relative abundance of liter species found in the channel. The amount of liter in the channel is a function of the channel configuration, the presence of retention structures, and the height of the litter producing vegetation. Forested stream channels that have been calibrated by known litter releases have retained most of the liter within 100 meters of the release point.]

8. Are there regional differences in the effects of natural disturbance or forest management activities on the biotic or nutrient riparian area functions?

a. Do the same disturbance regimes or management activities have different effects in different regions (e.g. the coastal coast range, interior coast range, Cascade, or Klamath - Sierra Nevada)?

[Explanation: this is based on the fact that there are significant geological, rainfall, and temperature regime differences from west to east, from low to high elevations, and from north to south.]

KC 4/17/07

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Appendix B: Wood Riparian Exchange Function

Primer, Key Questions and Initial List of Literature to be reviewed.

Primer
on
Wood
Riparian Exchanges Related to Forest Management in the Western U.S.

Prepared by the
Technical Advisory Committee
of the
California Board of Forestry and Fire Protection

May 2007

Version 1.0

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PRIMER: WOOD RIPARIAN EXCHANGE FUNCTION

(Abstracted from Hassan, Hogan, Bird, May, Gomi, and Campbell, Spatial and temporal dynamics of wood in headwater streams of the Pacific Northwest, Jour of the Amer Water Res Assn., Aug 2005.)

In general, wood within the channel boundary significantly alters flow hydraulics, regulates sediment transport and storage, and influences channel morphology and diversity of channel habitat (e.g., Swanson and Lienkaemper, 1978; Hogan, 1986; Bisson *et al.,* 1987; Montgomery *et al.,* 1995, 1996).

In-channel wood plays an important role in determining aquatic habitat conditions and riparian ecology (e.g., Bisson *et al.,* 1987; Bilby and Bisson, 1998).

Wood is introduced to the stream channel through a variety of processes including mass wasting, tree fall (blowdown), and bank erosion.

Fluvial and nonfluvial processes transport and redistribute wood introduced in upstream areas to downstream locations (e.g., Keller and Swanson, 1979; Lienkaemper and Swanson, 1987; Nakamura and Swanson, 1993; Hogan *et al.,* 1998; Johnson *et al.,* 2000a; Benda *et al.,* 2002, 2003; Lancaster *et al.,* 2003).

However, wood exerts its greatest geomorphic influence in channels with physical dimensions similar to or smaller than the size of wood (e.g., Bilby and Ward, 1989; Bilby and Bisson, 1998);
therefore, wood plays a disproportionately large role in small headwater streams.

Although wood dynamics and channel morphology of streams in the PNW have been studied in some detail, most of the research has occurred in relatively large streams and rivers (> third-order streams on 1:50,000-scale maps). Such results may not be applicable in headwater streams where episodic sediment and wood supply from adjacent hillslopes dominate channel dynamics and where fluvial transport of wood is restricted due to insufficient streamflow and narrow channels. The practical need to understand the physical and ecological roles of small streams has recently been highlighted by interest in restoring downstream ecosystems and the assessment of land management practices in relatively small watersheds (Moore and Richardson, 2003).

Interest in wood dynamics in headwater channels stems from the recognition that these channels represent a distinct class of stream, with characteristic morphologies, processes, and dynamics (see Benda *et al.,* 2005; Hassan *et al.,* 2005).

The focus is on the steeper portion of the channel network where episodic wood inputs and sediment from adjacent hillslopes exert significant control on channel dynamics and morphology. In these channels wood tends to accumulate, and sediment is stored upstream of accumulations, transforming steep bedrock channels into alluvial reaches (Massong and Montgomery, 2000; May and Gresswell, 2003b; Montgomery *et al.,*2003b).

In these streams, wood controls channel morphology by regulating the temporal, spatial character and the quantity of sediment stored within the channel zone, and this influences channel stability (e.g., Swanson *et al.,* 1982; Bilby and Ward, 1989).

The paper begins by defining small streams and addressing wood scaling issues relative to channel size. Then the paper reviews the current knowledge regarding each component of the wood budget in small streams. Next the paper discusses the spatial and temporal variability of wood in small streams, with special attention to geographic variability. Then an assessment of available models for the predicting wood dynamics in small streams is provided. The effect on wood dynamics of timber harvesting and riparian management on wood dynamics is considered. Finally, gaps in the knowledge are identified for future research on the wood dynamics in small streams. Due to the limited available information on small forested streams, certain information obtained from larger mountain rivers will be included in this review, and its applicability to small streams is assessed.

Table 1 – Definition of relative wood size and relative channel size. Matrix thresholds are arbitrary until further analysis justifies these classes. This scaling of wood to channel size allows use of studies in larger channels.

TABLE 2. Definition Matrix of Relative Wood Debris Size and Relative Channel Size.

	Relative LWD Size Ll/Wb			Relative Channel Size Ll/Wb		
Ld/Db	< 0.3	0.3-1.0	> 1.0	< 0.3	0.3-1.0	> 1.0
<0.3	S	M	L	Large	Intermediate	Small
0.3-1.0	M	L	L	Intermediate	Small	Small
> 1.0	L	L	VL	Small	Small	Very Small

Notes: Ll = log length; Ld = log diameter; Wb = channel bankfull width; Db = channel bankfull depth; S = small woody debris (SWD); M = intermediate wood debris (MWD); L = large woody debris (LWD); VL = very large organic debris; D = dominant grain size (~ D_{95}). D/Ld should be meaningful such that D/Ld: > 1 debris less important because bed material provides primary structural functionality; 0.3-1.0 debris more important and structurally functional; < 0.3 debris critically important.

Value of, need for, a wood budget to determine where wood comes from, where it is delivered to, where it is stored, how it is transported or depleted from a given drainage basin or stream reach.

From a forest management context there is potential to affect each component of the budget, so it is important to know the relative importance of each component and which are most susceptible to impact.

Wood Recruitment

The potential of landslides in mountainous landscapes can be increased by logging, road building, wind throw wildfire, earthquakes, and volcanic activity (Harmon *et al.,* 1986; Lienkaemper and Swanson, 1987; Nakamura and Swanson, 2003).

Research in the PNW has shown that landslides can provide a substantial quantity of wood to headwater streams (Keller and Swanson, 1979; Schwab, 1998; Hogan *et al.,* 1998; May, 2002; May and Gresswell, 2003a; Reeves *et al.,* 2003).

In contrast, other studies in Alaska, California, and Washington have found that mass movements may be of limited importance in supplying wood to larger streams (Murphy and Koski, 1989; Johnson *et al.,* 2000a; Martin and Benda, 2001; Benda *et al.,* 2002; Gomi *et al.,* 2004; May and Gresswell, 2004).

Another wood source into small streams is snow avalanches, a process that commonly destroys forest stands in the runout pathway. Repeated avalanches down established pathways prevent the growth of mature forests, so this process may be associated with the recruitment of relatively small wood. Where snow avalanches are an important landscape process, they provide the greatest wood recruitment in areas where the channel and hillslopes are coupled (Dave McClung, The University of British Columbia, January 6, 2005, personal communication) (see Figure 1 below)

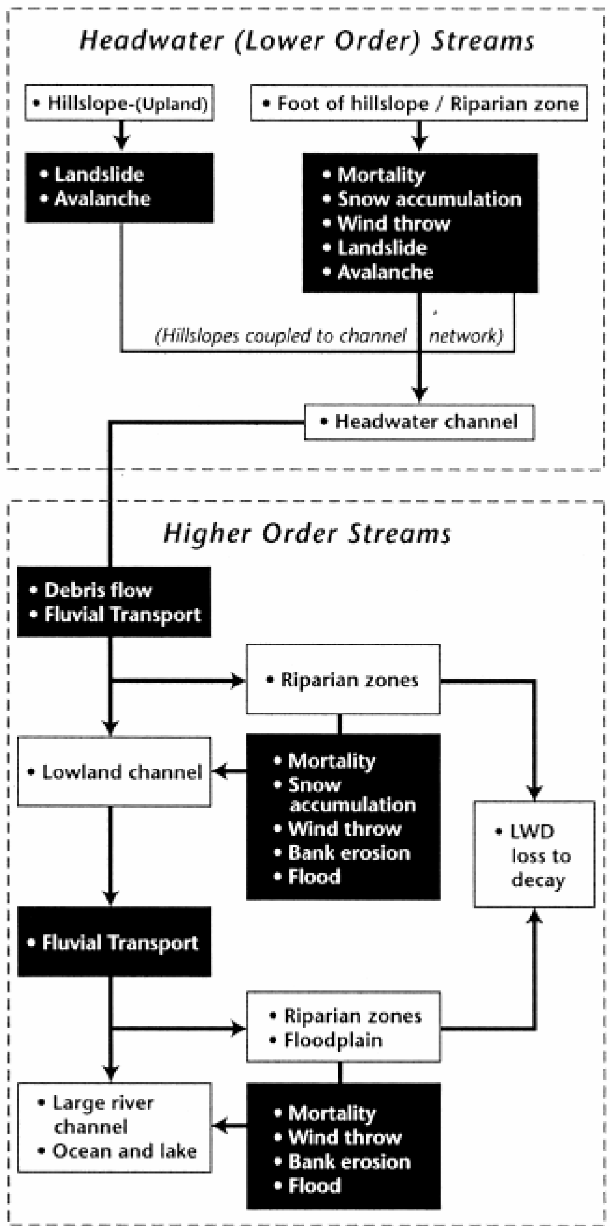

Figure 1. Flow Diagram for a Wood Budget in a Watershed. Open squares represent geomorphic areas related to locations for the sources and storages of wood, and filled squares represent processes that affect wood transport.

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Fires, insect infestations, and disease outbreaks are other processes that influence the recruitment of wood to streams.

If high severity fires burn extensive areas around headwater streams, the amounts and characteristics of wood input to streams may be altered for long periods; wood inputs are likely to increase immediately after fires (Nakamura and Swanson, 2003). Burned wood may also break into smaller pieces that can choke the channel, thereby increasing channel instability and downstream fluvial transport of wood (e.g., Berg *et al.,* 2002). The degree of fire damage to stands depends on fire severity, type (ground, surface, or crown), and spatial extent (Agee, 1993). Patterns of mortality due to forest fire vary among regional fire regimes, season, and topography.

Compared to floodplains, upland areas, including small streams and riparian zones, are more frequently affected by forest fires because of their relatively dry conditions and strong winds (Agee, 1993). Fire can also affect the wood budget by altering the age structure of the forest, initiating episodic pulses of wood recruitment, consuming existing dead wood, and influencing the mobility of instream wood (Young, 1994; Tinker and Knight, 2000; Zelt and Wohl, 2004).

Finally, insect infestations and disease outbreaks can episodically affect stand mortality in large areas. In the PNW, many disease and insect outbreaks appear to be related to fire suppression or exotic pathogens (Hessburg *et al.,* 1994; Swetnam *et al.,* 1995; Dwire and Kauffman, 2003). However, most insects and diseases affect only a single tree species, so the net effect on wood recruitment will depend upon the composition of the stand (Harmon *et al.,* 1986).

Streambank erosion may not significantly contribute wood to steep headwater streams because the channel is constrained by the adjacent hillslopes (Nakamura and Swanson, 2003) and banks are often semi- or non-alluvial (e.g., Halwas and Church, 2002). Actual rates of bank erosion in headwater constrained streams are poorly documented but are believed to be minimal. However, in gentler areas with less bedrock constraints, bank erosion is likely (expected) to be a significant source of wood into channels. In headwater streams, wood is often suspended above the channel banks due to relatively narrow channel widths (relative to tree heights and diameters) and hillslope confinement. Direct input to the channel may not occur until a log is either broken or fragmented (Nakamura and Swanson, 1993).

Wood storage

Once delivered to the stream system, wood is stored for various durations in several different nvironments; these include areas in riparian zones and associated floodplains and within the channel boundaries (Figure 1, Table 3).

few studies have referenced the criterion used to determine that portion of the wood actually interacting with the stream and fluvial processes. Robison and Beschta (1990a) examined the storage of wood in distinct zones within the stream system and developed a classification system in which they identified and distinguished between wood within the channel and wood on the banks.

Storage of wood within a system can be likened to a wood reservoir that has a characteristic residence time (Keller and Tally, 1979; Hogan, 1989). Wood reservoirs can be used to study wood dynamics over a range of temporal and spatial scales. In headwater streams, the temporal scale is likely to be a function of the frequency and magnitude of the wood mobilizing events (see the following section).

Wood output

Wood stored in the fluvial system is transferred out of a reach by downstream transport or lost through abrasion or *in-situ* decomposition.

Log stability in channels is controlled by many factors, including piece dimensions (length and diameter) relative to the channel, wood integrity, attached root wads, and degree of anchoring in the channel bed and bank (e.g., Montgomery *et al.,* 2003a,b).

Braudrick *et al.* (1997) suggested three mechanisms of wood transport: floating in a congested manner (high concentration) by streamflow, floating in an uncongested manner, and debris flows (for more details see the section on modeling).

Field studies show that log movement is more likely to occur as channel size increases and when logs are shorter than bankfull width, implying that fluvial transport of wood is more significant in higher order streams (e.g., Bilby and Bisson, 1998).

Wood temporal and spatial variability

Tthreshold occurs that corresponds to channels approximately 5 m wide, which is similar to the pattern observed by Jackson and Sturm (2002).

(Excerpted from Lassettre and Harris, 2002, The Geomorphic and Ecological Influence of Large Woody Debris in Streams and Rivers)

Timber harvest activities in streamside forests can directly affect wood input (Table 2, Swanson and Lienkaemper 1978, Bilby and Bisson 1998).

Table 2. The effect of certain management practices on the characteristics and abundance of LWD within stream systems. Timber harvest temporarily reduces input or changes the physical characteristics of subsequent inputs. Flood control and road maintenance activities generally result in the removal of in-channel wood.

MANAGEMENT PRACTICE	EFFECT	REFERENCES
Timber harvest	• Temporary reduction in LWD input	Bryant 1980, Andrus 1988, Murphy and Koski 1989
	• Second growth input smaller, less rot resistant with less profound effects on physical habitat	Bilby and Ward 1991, Wood-Smith and Buffington 1996, Ralph et al. 1994
	• Removal of logging residue simplifies physical habitat by failing to distinguish between naturally occurring habitat-forming logs and leftover material	Swanson et al. 1976, Swanson and Lienkaemper 1978, Beschta 1979, Bryant 1980, Keller and MacDonald 1983, Bilby 1984, Bisson et al. 1987, Bilby and Ward 1989
	• Extremely large amounts of logging material reduces intragravel flow, increases biological oxygen demand, reduces space available for invertebrates, and blocks fish migration	Hall and Lantz 1968, Narver 1970, Brown 1974
	• Destabilization of hillslopes and increase in debris avalanches	Swanson and Lienkaemper 1978
	• Narrow buffer strips (<20 m to 30 m) potentially reduce wood input	McDade et al. 1990, Van Sickle and Gregory 1990
	• Buffer strips adjacent to clearcuts have higher occurrence of windthrow and are depleted of large wood sources rapidly	Reid and Hilton 1998
Flood control and road maintenance	• Remove wood to decrease channel roughness, increase conveyance, and maintain flood capacity	Marzolf 1978, Young 1991, Gippel et al. 1996
	• Remove wood and clear jams to keep culverts and bridges free of debris and reduce structural damage during storms	Singer and Swanson 1983, Diehl 1997

The harvesting of streamside forests may temporarily reduce or eliminate LWD recruitment to the stream (Bryant 1980).

The recovery time for input to return to pre-harvest conditions may be quite long. Fifty years after logging, debris from the current stand of a western Oregon stream contributed only 14% of total LWD volume and only 7% of the wood from the current stand contributed to pool formation (Andrus et al. 1988).

The results indicate that some second growth stands must grow at least 50 years before trees contribute LWD in sizes and amounts similar to old growth forests. A decay model calibrated in southeastern Alaska predicted a 70% reduction in wood 90 years after clear-cutting, and that full recovery exceeded 250 years (Murphy and Koski 1989).

Streams flowing through second growth forests have a lower frequency of LWD associated pools and fewer channel spanning logs than old growth streams, leading to a scour pool dominated system (Bilby and Ward 1991). Thus, in low to mid-order streams the percentage of LWD formed waterfalls and the control of wood on gradient is decreased by timber harvest.

Old growth logs are larger and retain more bedload sediment and fine organic debris. Fine organic debris influences the physical characteristics of large jams and may contribute to an increased diversity of pool types in old growth streams (Bilby and Ward 1991).
Changes in wood loading and abundance significantly alter stream morphology. Wood-Smith and Buffington (1993) showed that pool frequency, pool depth, and local shear stress were significantly different in logged versus unlogged streams.

Near-stream logging influences natural LWD input processes. Depending on the method, harvest activities destabilize hillslopes and increase the likelihood of debris avalanches (Swanson and Lienkaemper 1978).

Buffer strips are a common technique to reduce logging effects on forests and streams. Most LWD inputs come from within 20 m to 30 m of the stream channel and buffers more narrow than this zone of input potentially reduce the amount of available logs (McDade et al. 1990, Van Sickle and Gregory 1990).

Buffer strips adjacent to clearcuts are exposed to higher wind velocities, increasing the occurrence of windthrown logs to the stream channel (Reid and Hilton 1998).

In moderate to high gradient streams, logs play an important role in bedload storage (Figure 2), and the removal of LWD eliminates potential storage sites (Beschta 1979, Bilby 1984, Bilby and Ward 1989).

The decrease in storage capacity and subsequent release of sediment simplifies physical habitat by filling in the deepest pools, reducing pool area, and smoothing channel gradient (Sullivan et al. 1987, Dominguez and Cederholm 2000).

Debris removal affects salmonid populations by decreasing the amount of available hydraulic cover available during winter high flows, and by reducing stream wetted width and perimeter (Dolloff 1986, Elliott 1986).

Alternatively, an excessive amount of logging material left in the stream may be damaging to fish populations. Fine debris lying on the gravel surface impedes interchange between intragravel flow and surface water, reducing subsurface dissolved oxygen levels (Hall and Lantz 1969, Narver 1970, Brown 1974).

Reduced oxygen availability retards the development of salmonid embryos within the gravel. The decomposition of wood increases biological oxygen demand, further reducing available dissolved oxygen (Narver 1970).

Small pieces of wood and bark occupy interstitial pores, reducing the available living space for stream invertebrates (Narver 1970).

Very large human induced accumulations of wood prevent upstream migration of anadromous salmonids (Brown 1974). Much historical management of LWD in logged streams concentrated on the removal of excess debris to allow fish passage (Bilby and Bisson 1998).

In systems influenced by human infrastructure, road maintenance and flood control activities affect the abundance of large wood. Logs and riparian vegetation increase channel roughness, reduce conveyance, and are commonly removed by managers to maintain flood capacity (Marzolf 1978, Singer and Swanson 1983, Young 1991, Gippel et al. 1996).
Possibly the first step in improving the management of LWD in California stream systems is to recognize the different roles it plays in different parts of the watershed. The stream classification proposed below explicitly does that.

Table 3. The gradient range and general characteristics of reach morphologies in alluvial channels (Data taken from Bisson and Montgomery 1996 and Montgomery and Buffington 1997).

	CASCADE	STEP-POOL	PLANE-BED	POOL RIFFLE
GRADIENT	• 0.08 to 0.30	• 0.04 to 0.08	• 0.01 to 0.04	• 0.001 to 0.02
BED MATERIAL	• Boulder	• Cobble/boulder	• Gravel/cobble	• Gravel
CONFINEMENT	• Confined	• Confined	• Variable	• Unconfined

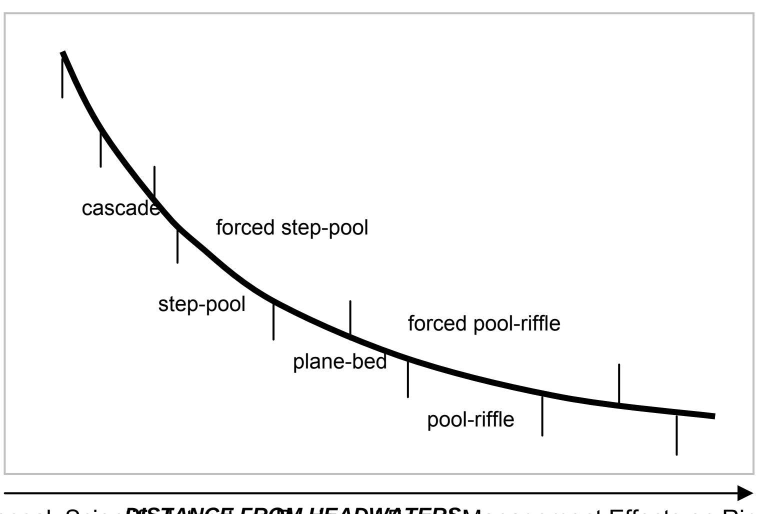

Figure 2. Generalized long profile of alluvial channels showing spatial arrangement of reach morphologies, including forced step-pool and forced pool-riffle morphologies. Forced morphologies extend beyond the gradient range of free-formed counterparts. Gradient ranges of forced morphologies depicted above are interpreted from Montgomery et al. (1995) and Beechie and Sibley (1997). The classifications are based on geomorphic processes and reflect basin wide trends in sediment transport and storage (Figure adapted from Montgomery and Buffington 1997).

To ensure future supplies of LWD to stream channels, buffer strips serving as reservoirs of wood supply should be wide enough to encompass the zone of LWD input, typically within 20 m to 30 m of the stream channel (Lienkaemper and Swanson 1987, McDade et al. 1990, Van Sickle and Gregory 1990).
Some researchers have argued for larger buffers, based on susceptibility of buffer strips next to clear-cuts to blow-down and rapid depletion of available streamside wood (Reid and Hilton 1998).

The use of a selectively logged fringe buffer adjacent to the streamside buffer may serve to reduce abnormally high rates of windthrow and preserve natural input rates. Any selective cutting within buffer strips should leave an abundant supply of the largest trees for recruitment (Murphy and Koski 1989, Abbe and Montgomery 1996).
The use of a selectively logged fringe buffer adjacent to the streamside buffer may serve to reduce abnormally high rates of windthrow and preserve natural input rates. Any selective cutting within buffer strips should leave an abundant supply of the largest trees for recruitment (Murphy and Koski 1989, Abbe and Montgomery 1996).

Species, diameter, and wood decay rates influence the amount of wood recruitment potentially necessary (Murphy and Koski 1989).

Along with the diameter and length of pieces of large wood, the riparian plant species involved largely determine the processing (turnover) time of large wood in streams. (e.g. Anderson et al. 1978; Anderson and Sedell 1979). The actual rate at which large wood of a given species is processed in a stream is a function of temperature, oxygen, moisture, microbial metabolism, invertebrate ingestion, and mechanical abrasion. Completely submerged wood is processed a great deal more slowly than damp wood , on which terrestrial fungal and invertebrate agents can act. (Harmon et al. 1986). In general, wood of hard wood species is processed more rapidly than that of coniferous species. For example, red alder is among the most rapidly and Douglas fir is among the slowest (Anderson et al. 1978). These differences is disappearance rates of the wood types are primarily dependent upon the relative activities of biological agents (microbes and invertebrates) on the wood (Harmon et al. 1986).

Table 4. The possible management implications of preserving LWD input, transport, and presence within the stream channel.

MANAGEMENT PRACTICE	IMPLICATION	REFERENCES
Timber harvest	• Buffer strips should be wider than zone of LWD input	McDade et al. 1991, Van Sickle and Gregory 1990
	• Fringe buffers can protect streamside buffers from premature wood depletion	Reid and Hilton 1998
	• Selective management in buffers should consider future input required based on instream surveys	Bilby and Ward 1989, Murphy and Koski 1989
	• Selective management should leave large trees that will be stable and influence channel morphology	Fetherston et al. 1995, Abbe and Montgomery 1996
	• Active management of buffer zones can increase recruitment of certain species and sizes of wood	Beechie and Sibley 1997
	• Removal of logging debris best dealt with by selective removal	Bryant 1983, Bilby 1984, Gurnell et al. 1995
	• Knowledge of habitat conditions, and the size and abundance of LWD required to maintain conditions must be considered when removing instream wood	Bryant 1983, Bilby 1984
	• Characteristics of unmanaged streams should guide re-introduction of wood	Smith et al. 1993a, b, Montgomery et al. 1995, Abbe and Montgomery 1996, Beechie and Sibley 1997, Montgomery and Buffington 1997
Flood control and road maintenance	• Must gain quantitative understanding of effect of wood on flood heights and how moves through a system	Young 1991, Braudrick et al. 1997, Braudrick and Grant 2000
	• Design and modify bridges and culverts to allow for passage of woody debris	Diehl 1997, Flanagan et al. 1998
	• Develop management that recognizes ecological value and impact of wood on human infrastructure and public safety	Singer and Swanson 1983, Piegay and Landon 1997

Forest managers should seek to increase the recruitment of certain species, primarily conifers which produce the largest and longest lasting LWD. This may involve active management of deciduous riparian zones to promote conifer establishment and growth (Beechie and Sibley 1997). This strategy should be considered in relation to position within the channel network. Small channels (<10 m width) can form pools around smaller pieces of wood (<20 cm), such as alder logs. Large to intermediate channels require greater diameter logs to form pools (>60 cm). Data on variations in the size and amount of woody debris with changing stream size could be used to develop plans for numbers and sizes of trees to be achieved (Bilby and Ward 1989).

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CJ 2/12/07

KEY QUESTIONS: WOOD

A significant body of literature exists documenting the relationship of forest management practices to *in-stream wood* recruitment, delivery, budgeting, and future production along riparian zones. Seeking to resolve the remaining uncertainties related to forest management effects on *in-stream wood* and the riparian zone is the emphasis of this investigation for the BOF TAC.

Embedded or implied in each key question below are the following issues for the contractor's synthesis:

A. Relationship to each of California's regions;

B. Context for riparian buffer strip size: stream order, stream class, topography, hydrologic regime, climate;

C. Context for comparisons: pristine, "optimum", legacy, or pre-harvest conditions;

D. Relationship of the quality of forest management practices being evaluated to current California forest practices; ability of BMPs to effectively mitigate identified problems;

E. Relationship of alterations to salmonid habitat quality and feeding effectiveness.

In the following question production of potential in-stream wood means the potential for living tree(s) in or near the riparian zone to become recruited as part of the dead and down wood in the stream.

1) How do forest management activities or disturbances in or near the riparian zone affect the production of potential *in-stream wood*, over space and time?

a.) To what extent is vegetation in or near the riparian zone surrounding lower order streams (e.g. 0, 1st, 2nd) a significant source of potential *in-stream wood* in unmanaged and managed forest areas? Do these results differ for larger order streams?

b.) What is the effect of current forest management practices, in or near riparian zones, bordering small and large order streams on production of potential *in-stream wood*? To what extent and in what ways does plant succession stage or vegetative community have an effect?

c.) To what extent and in what ways is production of potential *in-stream wood* from stream banks and flood-prone areas affected by current forest management practices?

d.) What characteristics of riparian buffer zones affect the production of potential *in-stream wood*? Is there a difference in wood production in unmanaged versus managed forests?

e.) What is the effect of current forest practices on incipiently available down wood in or near the riparian zone for in-stream wood production?

f.) How should forest management goals differ by stream order, vegetation type, and region to produce potential *in-stream wood* of the appropriate diameter size, species and other characteristics to maintain salmonid habitat over space and time? What minimum buffer widths have been shown to be effective?

g.) How can forest management practices encourage stand conditions that produce and maintain the potential for future *in-stream wood* over time?

h.) What is the effect of natural disturbance on the production of potentially available wood to the stream?

In the following question in-stream wood delivery means the physical process by which a living tree(s) became part of the dead and down wood in the stream.

2) How do forest management activities or disturbances in or near the riparian zone affect the delivery of *in-stream wood* onsite and/or downstream over space and time?

a.) To what extent and with what mechanisms are areas in or near riparian zones of lower order streams (e.g. 0, 1st, 2nd) a significant source of *in-stream wood* delivery in unmanaged and managed forest areas? How do these results differ for higher order streams? To what extent and with what mechanisms do low-order streams deliver *in-stream wood* to higher order, fish-bearing streams?

b.) To what extent and in what ways is *in-stream wood* delivery from stream banks and flood-prone areas affected by current forest management practices? To what extent and in what ways does plant succession stage or vegetative community have an effect?

c.) How does forest management affect *in-stream wood* delivery to channels?

d.) What is the effect of natural disturbance on the delivery of wood to a stream?

e.) What is the effect of stand-level riparian forest conditions on *wood* delivery to streams to maintain salmonid habitat?

f.) How should forest management goals differ by stream order, vegetation type, and region to deliver *wood to the stream* of the appropriate diameter size, species and other characteristics to maintain salmonid habitat over space and time? What minimum buffer widths have been shown to be effective?

3.) Based on the results of the above, what minimum buffer width and characteristics are shown to be needed to maintain production and delivery of wood to the stream from managed forests?

a.) How do these results vary by geographical region and process, size of watershed, stream order, forest species mix and age, stream reach, stream habitat present, forest practices within and nearby the riparian zone, fish species, etc.?

b.) How do these results vary by forest management practices in or near the riparian zone?

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APPENDIX C: HEAT RIPARIAN EXCHANGE FUNCTION –
KEY QUESTIONS, INITIAL LIST OF LITERATURE TO BE REVIEWED.

Primer
on
Heat
Riparian Exchanges Related to Forest Management in the Western U.S.

Prepared by the
Technical Advisory Committee
of the
California Board of Forestry and Fire Protection

June 2007

Version 1.0

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Prepared as background for the 2007 Scientific Literature Review of Forest Management Effects on Riparian Functions in Anadromous Salmonid Fishes for the California Board of Forestry and Fire Protection.

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PRIMER: HEAT RIPARIAN EXCHANGE FUNCTION: The Status of Knowledge for Heat Transfer Affecting Stream Temperature and Microclimate within Riparian Forest Buffers

This primer discusses the processes of heat transfer within riparian ecosystems and the effect of forest management on water temperature and microclimate. These interactions have been thoroughly and thoughtfully reviewed in a recent review article by R.D. Moore, D.L. Spittlehouse, and A. Story that appeared in the Journal of the American Watershed Resources Association (2005). This article was part of a compendium of review articles by leading researchers in the field. This review paper provides a very strong discussion of the mechanics of heat transfer and the role of riparian forests and stream factors in determining water temperature and microclimate characteristics in managed and unmanaged forest streams. The TAC adopts this review paper as the primary basis for the heat and microclimate primer.

> Moore, R. D, D.L. Spittlehouse, and A. Story. 2005. Riparian Microclimate and stream temperature response to forest harvesting: a review. Journal of the American Water Resources Association 41(4): 813-834.

The Moore et al. review paper (2005) was primarily focused on small streams, and does not thoroughly cover several topics important to the discussion of T&I rules in California. These include the effects of water temperature on salmon, and watershed-level temperature patterns. The TAC committee authored a discussion of these topics that reviews the scientific literature in some depth on these topics. These two documents together serve as the TAC's Primer on Heat Transfer and Microclimate in Riparian Areas. The TAC considers the literature reference lists attached to each of these two documents to be the supporting literature for the Primer. Because the fine print in the copy of the Moore et al. (2005) article included in this package may be difficult to read, we have reproduced a copy of the literature citations and included it behind the article.

Finally, the TAC developed a set of key questions that are meant to guide and focus the BOF literature review on the subject of riparian forests, heat transfer, microclimate, and salmon health. The TAC also has identified recent references that should serve as a core for that literature review.

TABLE OF CONTENTS

Summary of the effects of forest harvesting on riparian microclimate and stream temperature--a synthesis of key points from Moore et al. 2005 and the TAC Discussion of the Biological Effects of Temperature on Salmonids

This summary follows the organization of the Moore, Spittlehouse, and Story (2005) review of Temperature and Microclimate published in the Journal of the American Water Resources Association in 2005. Key points are taken from this paper and summarized here in bulletized form. A similar summary of the key points of the TAC-developed temperature biological effects and watershed temperature patterns is appended to this summary.

The bulletized points in this document faithfully summarize the key findings of the Moore et al.(2005) paper, and the TAC addendum. These concepts were developed with thorough referencing in the Moore et al. review article and the TAC primer. For ease of reading, little or no referencing is included in this summary. The reader is urged to read both documents provided after this summary.

Introduction

1) There have been many studies of stream temperature and somewhat fewer for riparian microclimate.

2) There have been some excellent reviews previously (e.g. Beschta et al 1987).

3) There is still a lively debate about how to manage riparian zones to protect temperature and microclimate.

4) Most States require a riparian buffer to protect stream temperature and microclimate.

5) The Moore et al review (2005) concentrates on small streams in the Pacific Northwest.

Riparian Microclimate

Characteristics of Forest Microclimates

1) Forest canopies affect the microclimate and ultimately stream temperature because canopies intercept the transmission of radiation.
2) Tree species and stand densities affect evaporation processes, wind and light transmission.
3) Riparian areas typically have elevated water tables and higher soil moisture than adjacent upland areas.
4) Forest canopies tend to reduce the diurnal air temperature range compared to open areas (also reduce the soil temperature range).
5) Lower air temperatures under a canopy will also create higher humidity as well.
6) Relationship of riparian forest stands to topography will influence the extent, climate within, and effect on streams.

Edge Effects and the Microclimate of Riparian Buffers

1) The magnitude of harvesting related changes in riparian microclimate will depend on the width of riparian buffers and how far edge effects extend into the buffer.

2) There have been studies of microclimate effects in forests, and to a more limited extent, riparian areas, around the world.

3) Much of the change in microclimate takes place within about 1 tree height (15 to 60 m) of the edge.

4) Solar radiation, wind speed, and soil temperature adjust to interior forest conditions more rapidly than do air temperature and relative humidity.

5) Edge orientation can be important, particularly when south facing.

6) Studies of microclimate in riparian areas are more limited. (Cites Ledwith from California: 1.6 deg C decrease in air temperature per 10 m of buffer up to 30 meters and 0.2 deg C per 10 m for widths from 30 m to 150 m.

7) Only one pre-harvest/post-harvest study (Washington). Gradients from stream into upland existed for all variables except solar radiation and windspeed. May have been enough to influence riparian fauna.

Thermal Processes and Headwater Stream Temperature

1) An understanding of thermal processes is required as a basis for understanding stream temperature dynamics, in particular for interpreting and generalizing from experimental studies of forestry influences.

2) As a parcel of water flows through a stream reach, its temperature is a function of energy and water exchanges across the water surface and the streambed and banks, and changes as energy inputs change.

3) The temperature of a parcel of water represents the net heat exchange by radiation, turbulent exchange with the air (evaporation and convection), and conduction across the water surface and stream bed. If additional water is advected into the reach from groundwater or hyporheic exchange, the temperature of the parcel will also be determined by the volumetric mixing of the temperature of the incoming water (Figure 1).

   Energy is transferred to the stream and the surrounding environment by solar radiation. Energy is exchanged between the stream and the sky and atmosphere, the vegetation and/or surrounding topography, and the streambed. The potential for transferring heat among water, air, and vegetation is driven by the temperature gradients between them and the properties of each that determine how well each material transmits energy or conducts heat.

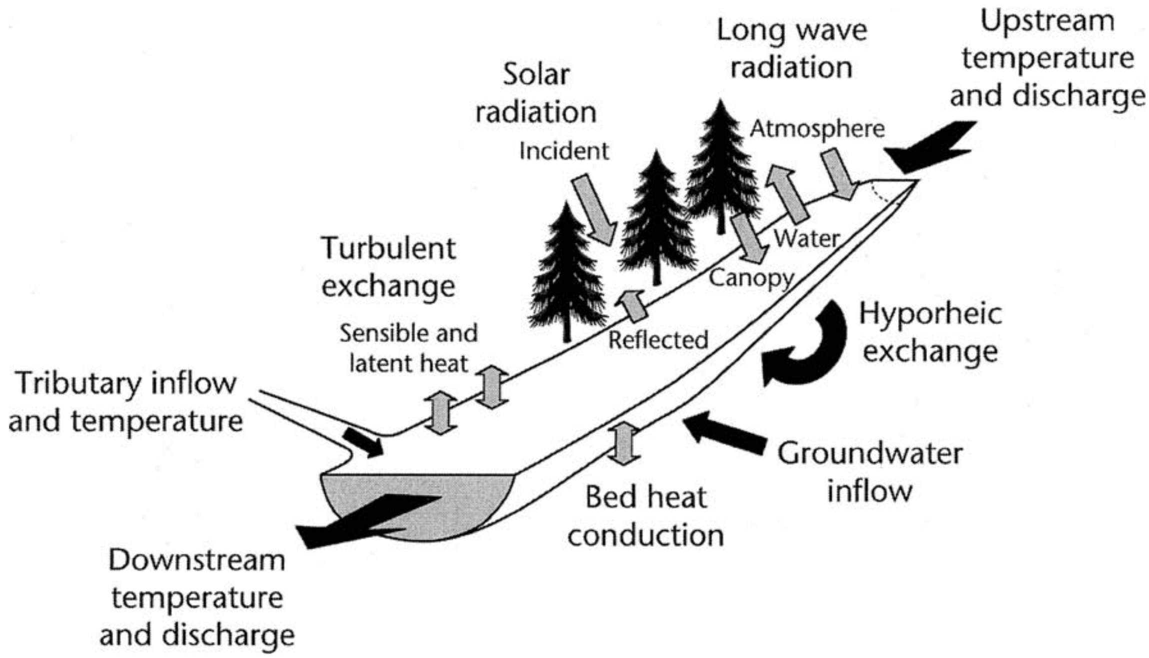
   Radiation inputs to a stream surface include incoming solar radiation (direct and diffuse) and longwave radiation emitted by the atmosphere, forest canopy and topography.

   Energy is exchanged between the water and air via convection (the transfer of heat from a surface to a moving fluid) and by evaporation. These processes are driven by wind speed and the vapor pressure and temperature of air.

   Energy is exchanged between the water and streambed via conduction.

4) A form of a reach energy balance equation is provided (Refer to Moore et al. 2005).

Figure 1. Factors controlling stream temperature. Energy fluxes associated with water exchanges are shown as black arrows. (From Moore et al. 2005).



Radiative Exchanges

1) Radiation inputs to stream surface include incoming solar radiation (direct and diffuse) and long-wave radiation emitted by the atmosphere, forest canopy and topography.

2) Canopy will reduce the direct component of solar radiation and will redistribute some of the diffuse component. The details of solar radiation transmission through canopies are complex because of the complexities of the vegetation surfaces and materials and the horizontal and vertical variation in canopy density.

3) Channel morphology (wide, narrow, and topographically shaded) will influence how much energy exchange will be blocked by vegetation or topography. Stream orientation relative to the path of the sun can also affect how long the stream "sees" the sky during the day.

4) When direct radiation comes from +30 degrees above the horizon, most of it can be absorbed within the water column and by the bed, and thus is effective at stream heating. Vegetation or topography must block radiation within this sector of the sky view to be effective.

5) Low solar angles at dawn and dusk, and during much of the annual solar cycle are not effective at stream heating because direct radiation comes in at too low of an angle to be absorbed and is reflected. Vegetation within this sector of the sky view is not important in shading the stream.

6) Incoming longwave radiation will be a weighted sum of the emitted radiation from the atmosphere, surrounding terrain, and the canopy, with the weights being their respective view factors.

7) Peak daytime net radiation over a stream without sky view blocking from canopy or topography can be more than five times greater than that under a forest canopy during summer.

Sensible and Latent Heat Exchanges

1) Energy is transferred between the water and the air by evaporation and convective heat transfer processes. Convection involves heat transfer between a fluid and an adjacent surface (air). Evaporation involves the transfer of heat energy with a mass of water to the air. Convective heat and mass transfer both depend strongly on the development of an aerodynamic boundary layer so they are strongly correlated to each other.
2) Natural convection is due to the motion of the fluid due to the temperature, and therefore density, differences at the surface and away from the surface. Forced convection is due to the movement of the fluid due to external forces such as wind. The rate at which energy is transferred by convection depends on the temperature difference between the water surface and overlying air, the wind speed, and the thermal conductivity of the air.
3) Evaporation depends on these factors, as well as the evaporative mass transfer coefficient as a function of wind speed.
4) Where the stream is warmer than the air, heat transfer away from the stream is promoted by the unstable temperature stratification. Where the air is warmer than the stream, the heat transfer from the air to the stream is dampened by the stable air temperature stratification.
5) Heat loss via evaporation can be a particularly effective dissipation mechanism at higher water temperatures for larger streams.
6) Heat energy exchange over very small stream may be limited by bank sheltering, particularly for narrow incised streams, potentially damping the

effects of openness to the sky.

Bed Heat Exchanges and Thermal Regime of the Streambed

1) Radiative energy absorbed at the streambed may be transferred to the water column by conduction and turbulent exchange and into the bed sediments directly by conduction and indirectly by advection in locations where water infiltrates into the bed. Given that turbulent exchange is more effective at transferring heat than conduction, much of the energy absorbed at the bed is transferred into the water column, and the temperature at the surface of the bed will generally be close to the temperature of the water column, except where there may be local advection of water with a different temperature.

2) Energy is also transferred between the water and streambed by conduction (the transfer of energy at the molecular level). The direction and rate of transfer depends on the temperature gradients within the bed and the thermal conductivity properties of the bed material.

3) The bed will normally serve as a heat sink and thus act as a cooling influence on the water on summer days. At night the bed transfers heat back to the water, serving as a warming influence. The net effect is to reduce the diurnal temperature range.

4) Bed materials have different thermal conductivity. Bedrock is very effective at absorbing heat, while pebbly surfaces are less effective.

5) There is a thermal gradient within a streambed from surface to depth. The temperature of bed surface sediments will be reasonably close to water temperature, and will experience daily fluctuation along with the stream water.

6) Increase in water temperature from forest management effects can translate into the bed for some distance, depending on the type of bed materials and temperature of the surface water and on the local hydrologic environment. The low thermal diffusivity of the stationary bed prevents extensive transfer of heat downward so that daily temperature variations diminish as depth increases. Daily temperature variation diminishes significantly by 0.5 meters.

7) The decrease in bed temperature with depth is what allows water that downwells into the streambed to cool.

8) Bed temperatures may be important biologically. The temperature influences the incubation environment of salmonids and the conditions for benthic invertebrates.

Groundwater Inflow

1) Groundwater is typically cooler than the stream's daytime temperature and warmer during winter, and thus tends to moderate diurnal and seasonal temperature variation
2) Forest harvesting can increase soil moisture and groundwater levels
3) Increases in groundwater volume could act to promote cooling, or at least ameliorate warming.
4) Some have argued cutting could increase groundwater temperature due to greater flow volume with decreased interception losses and transpiration.
5) There is no published research [at the time of this paper] that has examined groundwater discharge and temperature both before and after harvest as a direct test of the hypothesis of groundwater warming.

Hyporheic Exchange

1) Hyporheic exchange is a two-way transfer of water between a stream and its saturated sediments in the bed and riparian zone.

2) Stream water typically flows into the bed at the top of a riffle and re-emerges at the bottom of a riffle. If the temperature of hyporheic water discharging into a stream differs from stream temperature, then hyporheic exchange can influence stream temperature proportional to its volume and temperature.

3) Hyporheic exchange can create local thermal heterogeneity and it can be important for creating microhabitat characteristics of water temperature in relation to both local and reach scale temperature patterns in headwater streams.

4) There are significant methodological problems associated with quantifying rates of hyporheic exchange and its influence on stream temperature.

Tributary Inflow

The effects of tributary inflow depend on the temperature difference between inflow and stream temperatures and on the relative contribution to discharge and can be characterized by a simple mixing equation.

$$T_m = f_1T_1 + f_2T_2$$

where T is the inflow temperature and f is the proportional volume of the water bodies that join.

Longitudinal Dispersion and Effects of Pools

1) Longitudinal dispersion results from variation in velocity through the cross-section of a stream. Any effects on temperature distribution have not been well studied, but could smooth and dampen effects downstream.

2) Deeper pools may have incomplete mixing creating thermal stratification.

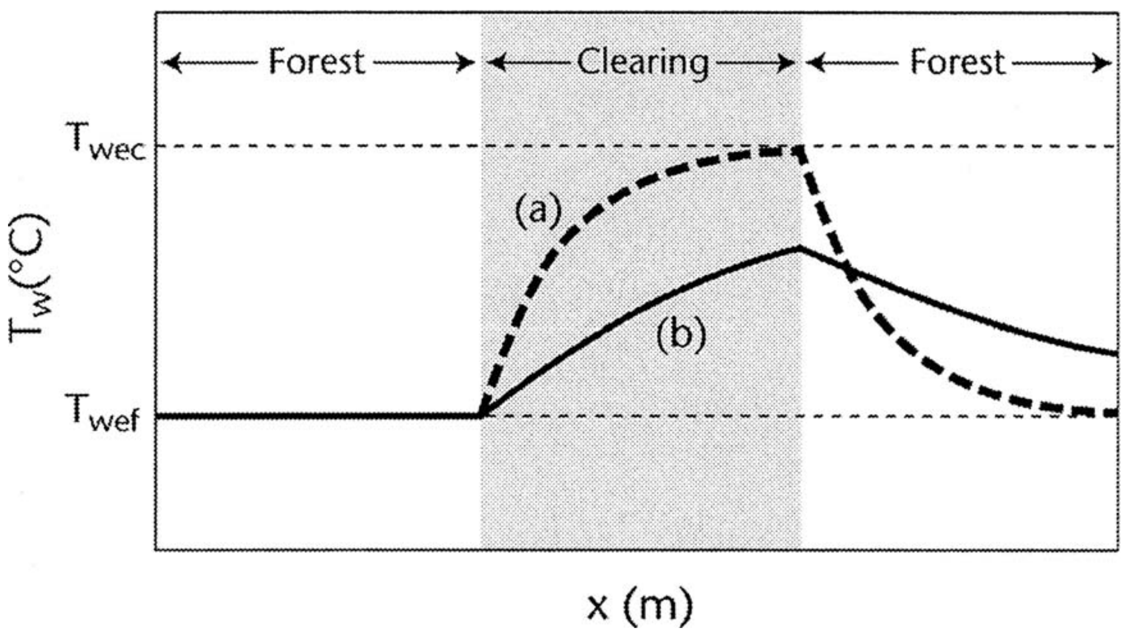
Equilibrium Temperature and Adjustment to Changes in Thermal Environment

1) For a given set of boundary conditions (e.g., solar radiation, air temperature, humidity, wind speed) there will be an “equilibrium” water temperature that will produce a net energy exchange of zero and thus no further change in temperature as water flows downstream.
2) There is a maximum possible temperature a parcel of water can achieve as it flows through a reach at a given time, assuming that boundary conditions remain constant in time and space.
3) The thermal environment changes spatially with new representative conditions in important driving environmental variables such as stream width, flow volume, view factor. The thermal environment changes in time with the daily and annual solar cycle. Changes in conditions will cause changes in the maximum temperature.
4) Equilibrium temperature may not be achieved because the boundary conditions may change in time and space before the water parcel can adjust fully to each thermal environment. A natural factor potentially limiting the downstream distance of thermal effects in small streams is the daily fluctuation of temperature with the solar cycle. Effects experienced in an upstream reach may be lost downstream as the stream cools at night.
5) Equilibrium temperature will be lower where there is substantial inflow of cooler groundwater and will be higher for unshaded reaches due to solar input.
6) The rate at which a parcel of water adjusts to a change in the thermal environment depends on stream depth because for deeper streams, heat would be added to or drawn from a greater volume of water per unit area. The deeper the stream, the less the diurnal fluctuation at the same solar input because of the thermal inertia of the water.
7) The temperature in shallow streams adjusts quickly to a change in thermal environment and solar radiation.
8) Flow velocity influences the length of time the parcel of water is exposed to a specific thermal environment. The speed with which the water parcel moves

determines whether it can adjust fully to that thermal environment before it passes into a new one.

9) Given that the depth and velocity of a stream tend to increase with discharge, the sensitivity of stream temperature to a given set of energy inputs should increase as discharge decreases.

Figure 2. Schematic temperature patterns along a stream flowing from intact forest, through a clear-cut, and back under intact forest for (a) shallow, low velocity and (b) deep, high velocity conditions. (Twef = equilibrium temperature in forest; Twec = equilibrium temperature in clearing) (From Moore et al. 2005).



Thermal Trends and Heterogeneity Within Stream Networks

1) Small streams tend to be colder and exhibit less diurnal variability when shaded than larger downstream reaches... Small streams tend to be more heavily shaded, often have a higher ratio of groundwater inflow, and are often located at higher elevations (cooler air).

2) Local deviations from a dominant downstream warming trend may occur as a result of ground water inflow, hyporheic exchange, advection of water from other sources, or even changes in dominant variables such as air temperature.

3) Thermal heterogeneity has been documented at a range of spatial scales: within a pool, within a stream reach, within a river system.

Stream Temperature Response to Forest Management

1) Many studies of the effects of forest management on stream temperature have occurred.

2) Some have BACI experimental design, some do not.

3) Most studies have been conducted in the PNW in rain-dominated climates.

Influences of Forest Harvesting Without Riparian Buffers

1) Almost all streams that have buffers removed increase in summertime temperature.

2) Harsh treatment yields high temperature response.

3) Results appear to be more mixed in more recent years with changes in forest practices that limit forest management in the riparian area.

4) Response in snowmelt-dominated areas is not well studied. However, there may be similar increases in stream temperature with canopy removal.

5) Winter temperatures have also not been well studied.

Influences of Forest Harvesting With Riparian Buffers

1) Studies in rain-dominated catchments suggest that buffers may reduce, but not entirely protect against increases in summer stream temperature. However, temperature increase is generally more moderate or very small when a buffer is left.

2) Two studies in snow-dominated areas in Canada have also shown an increase in temperature with complete canopy removal of 1 to more than $5^{o}C$ for a set of streams subject to a range of forest management treatments.

3) The protective effect of buffers can be compromised by blow-down.

Thermal Recovery Through Time

1) Post-harvest temperatures should decrease through time as riparian vegetation recovers.

2) Shade levels recover more rapidly in wetter forest types and at lower elevations.

3) Effects seem to last 5-10 years if riparian vegetation is allowed to recover.

4) Riparian canopy recovered more slowly when debris flows and channel disturbances affected streamside vegetation.

5) A study in subboreal B.C. suggested that shading by low vegetation may not be as effective at protecting water temperature as that from trees.

Comparison With Studies Outside The Pacific Northwest

1) Studies conducted elsewhere in the world are in many ways consistent with results from the PNW as dictated by the physics of heat transfer.

2) However, differences in important environmental variables, experimental techniques, and forestry practices limit the comparability of results.

Effects of Forest Roads

1) Some evidence for very small streams that even a road-right-of-way cut can be of sufficient length to cause local heating.

Downstream and Cumulative Effects

1) There can be a watershed level response to forest management, including a direct effect in disturbed reaches and by an upstream to downstream translation of temperature.

2) Downstream transmission of heated water would increase the spatial extent of warmer temperatures.

3) There is a debate about whether down-stream cooling (how much, how fast) can have a significant effect. Some studies show cooling or heating, while others do not.

4) Streams can cool in the downstream direction by dissipation of heat out of the water column via convection and evaporation, or via dilution by cool inflows.

5) Reported downstream temperature changes below forest clearings are highly variable. Some report streams cooled, some report streams continued to warm in the downstream direction.

6) Whether cooling occurs may depend on ambient air temperatures and hydrologic conditions within the downstream reach

7) To understand the mechanisms that allow cooling to occur requires more physical process-based research.

8) Three factors may mitigate against cumulative effects of stream warming. 1) dilution could mitigate temperatures to a biologically suitable level, 2) the effects of energy inputs are not linearly additive throughout a stream network due to systematic changes in balance of energy transfer mechanisms. 3) Intercepting environments (lakes, reservoirs).

9) There may be secondary impacts from forest management such as stream widening and shallowing that may occur with excess sedimentation that may change the heat exchange dynamics and influence water temperature.

Monitoring and Predicting Stream Temperature and its Causal Factors

Monitoring Stream Temperature

1) Most recent studies have used submersible temperature loggers to measure water temperature.

2) Forward-looking infrared radiometery from helicopters has been used to map the spatial distribution of temperature for investigating stream temperature patterns in medium to large streams. The application of this technology to small streams is limited. The method can identify cool water areas within larger rivers.

Measuring Shade

1) To account for riparian vegetation effects on temperature, there must be a measure of the extent to which the overstream vegetation blocks energy exchange with the water in the stream. Some type of measurement of canopy density is important, because this is the primary mechanism by which forest management affects water temperature.
2) Shade, canopy cover, canopy density, and view-to-the-sky are often variously used to describe or infer the effect of the riparian vegetation on water temperature. These measures express canopy as a density or percent overhead cover. However, these measures are not synonymous and will give different results when comparing riparian canopy cover among studies.
3) The vertical and horizontal variation in canopy characteristics that influence energy exchange are complex depending on the canopy structure and are variable along a stream reach. All measures must ultimately reflect an average condition that represents the thermal reach.

4) There are not only many different ways to describe the blocking influence of riparian vegetation but also many methods and measurement tools to estimate it.

   A. Blocking of the stream's total view to the sky: (yields measure of % openness or its inverse blockage)
      --Ocular estimates of the hemispherical view-to-the-sky aided by spherical densiometer or fisheye lens photography

B. Focused measurement of the area of the sky view through which the sun passes (yields measure of blocking of direct solar radiation primarily)

–geometric calculations based on canopy and terrain angles

--Spherical densiometer-type instrument modified to view the solar pathway

C. Indirect methods

--Compare radiation or lights levels using photovoltaic light meter above under the canopy and in the open

-- Back calculate canopy cover factor in heat energy balance by comparing temperature in open and under canopy

5) To date, there has been only moderate success with using the more complex or indirect measures.

Predicting the Influences of Forest Harvesting on Stream Temperature

1) Several authors have devised empirical models based on multiple regression from environmental variables to predict a selected temperature characteristic such as MWAT or MMWT. These types of models are simple with low requirements for input data, but they involve significant uncertainties, especially when applied to situations different from those represented in the calibration data. Nevertheless, several authors have developed locally relevant models that can usually predict maximum temperature within 2^{o}C with a regression coefficent (R^2) of 0.60 to 0.70)

2) The physics of heat transfer have been well studied, and a number of physically-based models incorporating energy balance concepts have been developed for application to individual stream reaches. These include the seminal model introduced by Brown (1969, 1985), TEMP-84 (Beschta and Wetherred, 1984), TEMPEST (Adams and Sullivan, 1989), Heat Source (Boyd 1996) and STREAMLINE (Rutherford et al. 1997)..

3) Physically-based models all work on the same physics principles but are constructed with with somewhat different assumptions, formulations, variables to inform, and complexity of environmental characterization.

4) There are also models to simulate stream temperatures at the stream network or catchment scale. These include SNTEMP (Mattax and Quigley, 1989), Bartholow (1991 and 2000), and a model based on the S=HSPF (Hydrological Simulation Program-FORTRAN) model developed by the U.S. Environmental Protection Agency and the U.S. Geological Survey (Chen et al 1998a, b).

5) Sullivan et al (1990) tested the ability of four reach scale models (TEMP-86, TEMPEST, Brown's model, SSTEMP) and three catchment scale models (QUAL2E, SNTEMP, and MODEL-Y) to predict forest-related temperature

increases in Washington. The reach models consistently (though not universally) achieved accurate temperature predictions (within 1-2°C) in many different types of streams and rivers. This was despite significant variability in the data required by the models and methods of measurement, especially with regard to riparian canopy. Simple models with relatively few variables performed as well as those that parameterized the environmental characteristics that drive the heat transfer modes in great detail.

6) The catchment scale models required more input data than would generally be available for operational applications foreseen by the Washington study, and did not provide accurate predictions for mean, minimum, and maximum temperatures as tested.

Discussion and Conclusions (From Moore et al. 2005)

Summary of Forest Harvesting Effects on Microclimate and Stream Temperature on Small Streams

1) Forest harvesting can increase solar radiation in the riparian zone as well as wind speed and exposure to air advected from clearings, typically causing increases in summertime air, soil, and stream temperatures and decreases in relative humidity
2) Riparian buffers can help minimize these changes
3) Edge effects penetrating into a buffer generally decline rapidly within about one tree height into the forest under most circumstances
4) Solar radiation, soil temperature, and wind speed appear to adjust to forest conditions more rapidly than air temperature and relative humidity.
5) Clearcut harvesting can produce significant daytime increases in stream temperature during summer, driven primarily by the increased solar radiation associated with decreased canopy cover but also influenced by channel morphology and stream hydrology.
6) Winter temperature changes have not been as well documented but appear to be smaller in magnitude and sometimes opposite in direction in rain-dominated catchments.
7) Although retention of riparian vegetation can help protect against temperature changes, substantial warming has been observed in streams with both unthinned and partial retention buffers.
8) Comparing results has been hampered by inconsistency in temperature metrics used among studies.

9) Increases stream temperatures associated with forest harvesting appear to decline to pre-logging levels within five to ten years in many cases, though thermal recovery can take longer in others. There is mixed evidence for the efficacy of low, shrubby vegetation in promoting recovery.

10) Temperature increases in headwater streams are unlikely to produce substantial changes in the temperatures of larger streams into which they flow, unless the total inflow of clearcut heated tributaries constitutes a significant proportion of the total flow of the receiving streams.

11) Streams heated by canopy removal may or may not cool when they flow into shaded areas. Where downstream cooling does not occur rapidly, the spatial extent of the thermal impacts is effectively extended to lower reaches, which may be fish bearing. In addition, warming of headwater streams could reduce the local cooling effect where they flow into larger streams, thus diminishing the value of those cool water areas as thermal refugia.

Biological Consequences and Implication for Forest Practices

1) It is difficult to estimate the biological consequences of harvesting related changes in riparian microclimate and stream temperature of small streams based on existing results.

2) In terms of terrestrial ecology in riparian zones, there is incomplete knowledge regarding the numbers of species that are unique to small streams and their riparian zones, as well as their population dynamics, sensitivity to microclimatic changes, and ability to recolonize disturbed habitat.

3) A better understanding is required of how changes in the physical conditions in small streams and their interactions with chemical and biological processes influence their downstream exports.

4) Based on the available studies, a one-tree-height buffer on each side of a stream should be reasonably effective in reducing harvesting impacts on both riparian microclimate and stream temperature.

5) Narrower buffers may provide at least partial protection, but their effectiveness may be compromised by windthrow. Alternative methods of designing buffers for protecting temperature in small streams may be explored.

Issues For Future Research (Moore et al. 2005)

1) Riparian microclimates have been relatively little studied, both in general and

specifically in relation to the effects of forest practices.

2) Shade is the dominant control on forestry-related stream warming in small streams.

3) Determining shade in small streams is difficult and refined and consistent methods are needed.

4) Hemispherical photography might be the way to go to solve subjectivity and methods problems.

5) The effects of low and deciduous vegetation in controlling temperature in very small streams is not well understood.

6) Further research should address the thermal implications of surface/subsurface hydrologic interactions, considering both local and reach scale effects of heat exchange associated with hyporheic flow paths.

7) Bed temperature patterns in small streams and their relation to stream temperature should be researched in relation the effects on benthic invertebrates and other aquatic species.

8) The hypothesis that warming of shallow ground water in clearcuts can contribute to stream warming should be addressed, ideally by a combination of experimental and process/modeling studies.

9) The physical basis for temperature changes downstream of clearings needs to be clarified. Are there diagnostic site factors that can predict reaches where cooling will occur? Such information could assist in the identification of thermal recovery reaches to limit the downstream propagation of stream warming. It could also help identify areas within a cut block where shade from a retention patch would have the greatest influence.

Summary Points of the TAC Discussion of The Effects of Temperature on Salmonids and Watershed Temperature Patterns

The Physiological Basis for Salmonid Temperature Response

1) Water temperature governs the basic physiological functions of salmonids and is an important habitat factor.
2) Fish have ranges of temperature wherein all of these functions operate normally contributing to their health and reproductive success. Outside of the range, these functions may be partially or fully impaired, manifesting in a variety of internal and externally visible symptoms. Salmon have a number of physiologic and behavioral mechanisms that enable them to resist adverse effects of temporary excursions into temperatures that are outside of their preferred or optimal range. However, high or low temperatures of sufficient magnitude, if exceeded for sufficient duration, can exceed their ability to adapt physiologically or behaviorally.
3) Salmon are adapted over some evolutionary time frame to the prevailing water temperatures in their natural range of occurrence, and climatic gradient are among the primary factors that determine the extent of a species' geographic distribution on the continent.
4) Salmon are considered a "cold water" species, and generally function best within the range of ambient temperatures in water bodies within their natural range of occurrence. This range is 0-30^{o}C for salmonids, where end temperatures are lethal and mid range temperatures are optimal. The southern limit of the natural range of salmonids coincides with the occurrence of summer water temperatures of 30^{o}C.
5) The effects of temperature are a function of magnitude and duration of exposure. Exposure to temperatures above 24^{o}C of sufficient continuous duration can cause mortality.
6) Salmon can tolerate each successively lower temperature for exponentially increasing intervals of time. Temperatures above 22^{o}C are stressful. Lengthy exposure to higher temperatures include loss of appetite and failure to gain weight, competitive pressure and displacement by other species better adapted to prevailing temperatures, or disease.
7) Growth occurs best when temperatures are moderate and food supplies are adequate. High and low temperatures limit growth. Optimal temperatures for growth are in the range of 14 to 17^{o}C, depending on species.

8) Salmon have been shown to increase in size in streams where riparian canopy was removed due to increased light and food availability, despite the occurrence of warmer temperatures.

9) Larger size generally increases survival and reproductive success.

10) Growth rates are important for anadromous salmonids, who must reach minimum sizes before they are able to migrate to the ocean. Missing normal migration windows by being too small or too large may have negative effects on success in reaching the ocean.

11) The temperature of rivers and streams ranges over the full range of temperatures within the range utilized by salmonids during the course of the year. The summer maximum temperatures are generally those of most concern.

12) The most thermally tolerant salmonid species that occur in California (steelhead, chinook and coho). Of these species, coho are the most thermally sensitive.

Temperature Exposure in Natural Streams and Potential Effects of Forest Practices

1) Water temperature generally tends to increase in the downstream direction with stream size as a result of systematic changes in the important environmental variables that control water temperature. As streams widen, riparian canopy provides less and shade until some point in a river system where it provides no significant blocking effect. Cooler groundwater inflow also diminishes in proportion to the volume of flow in larger streams.

2) The lowest order streams have the coolest water temperatures near groundwater temperature (11-14^{o}C). Higher order streams are near ambient air temperatures (20-26^{o}C). The range of water temperature from lower to higher orders in California rivers and streams during the warmest period in the summer spans much of the tolerable temperature range for salmonids. Water temperature typical of higher order streams are within stressful levels for salmonids.

3) Removal of riparian vegetation may increase stream temperatures up to the ambient air temperature, depending on the natural extent of shading and the proportion of canopy removed. Thus, temperatures typically observed only in downstream reaches may occur in tributary streams.

4) Salmonid distribution within stream systems and within the region reflects

temperature tolerance. Coho are found in the cooler waters associated with headwater streams and within the coastal zone where climate is strongly influenced by the Pacific Ocean. Steelhead have somewhat higher thermal tolerance, and are more widely distributed.

Copy of Journal Article by Moore, R. D, D.L. Spittlehouse, and A. Story (2005)

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RIPARIAN MICROCLIMATE AND STREAM TEMPERATURE RESPONSE TO FOREST HARVESTING: A REVIEW[1]

R. Dan Moore, D. L. Spittlehouse, and Anthony Story[2]

ABSTRACT: Forest harvesting can increase solar radiation in the riparian zone as well as wind speed and exposure to air advected from clearings, typically causing increases in summertime air, soil, and stream temperatures and decreases in relative humidity. Stream temperature increases following forest harvesting are primarily controlled by changes in insolation but also depend on stream hydrology and channel morphology. Stream temperatures recovered to pre-harvest levels within 10 years in many studies but took longer in others. Leaving riparian buffers can decrease the magnitude of stream temperature increases and changes to riparian microclimate, but substantial warming has been observed for streams within both unthinned and partial retention buffers. A range of studies has demonstrated that streams may or may not cool after flowing from clearings into shaded environments, and further research is required in relation to the factors controlling downstream cooling. Further research is also required on riparian microclimate and its responses to harvesting, the influences of surface/subsurface water exchange on stream and bed temperature regimes, biological implications of temperature changes in headwater streams (both on site and downstream), and methods for quantifying shade and its influence on radiation inputs to streams and riparian zones.
(KEY TERMS: stream temperature; forestry; headwater; riparian; microclimate; water quality; watershed management; Pacific Northwest.)

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INTRODUCTION

Riparian microclimate and stream temperature are critical factors in relation to habitat conditions in and near streams and are governed by the interactions of energy and water exchanges within the riparian zone. Riparian microclimate sets the boundary conditions for many of the energy exchanges that influence stream temperature, while stream temperature sets one of the boundary conditions for riparian microclimate. The two topics are therefore closely linked and are covered together in this paper, which focuses on research relevant to two concerns: (1) forest harvesting may change riparian microclimate and have an impact on aquatic and terrestrial habitat; and (2) forest harvesting, particularly with removal of riparian vegetation, may result in stream heating or other changes in water temperature that could have deleterious effects on aquatic organisms.

Despite decades of research on stream temperature response to forest harvesting, there are still vigorous debates in the Pacific Northwest about the thermal impacts of forestry and how to manage them (e.g., Larson and Larson, 1996; Beschta, 1997; Ice *et al.*, 2004; Johnson, 2004). The conventional approach to minimizing the effects of forest harvesting on streams and their riparian zones is to retain a forested buffer strip along the stream. Most jurisdictions in the Pacific Northwest require buffer strips to be left along larger (usually fish bearing) streams (Young, 2000). However, less protection is afforded to smaller, non-fish-bearing streams. For example, in British Columbia, buffer strips are not required along non-fish bearing streams unless they are a designated community water supply, and buffer strips are not mandatory along the fish bearing streams whose

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bankfull width is less than 1.5 m. Thus, small streams are potentially subject to significant changes in riparian microclimate and particularly to increased solar radiation, which is the major factor driving summertime stream warming.

Beschta *et al.*, (1987) presented an excellent review of the physical and biological aspects of stream temperature in a forestry context, but more recent research has expanded the geographic scope of knowledge within the Pacific Northwest (PNW) region, shed new light on governing processes, or made advances in relation to tools for monitoring and prediction. In the interests of completeness, this paper will revisit much of the material reviewed by Beschta *et al.* (1987) in addition to reviewing more recent studies but will focus on physical aspects. It is assumed that the reader has a basic grounding in microclimatological principles and terminology. Readers lacking this background are referred to Oke (1987) for an excellent introductory treatment.

Given that the primary concern is with riparian management around small streams, the review focuses as much as possible on studies in catchments less than 100 ha in area or streams less than 2 to 3 m wide. It also focuses on studies in the Pacific Northwest region, broadly defined to include northern California, Oregon, Washington, British Columbia, and southeastern Alaska. However, studies from outside the PNW region were considered if they provided useful insights that were not available from local studies. Similarly, studies that did not focus specifically on small forest streams were included if the results were relevant to small stream thermal regimes.

RIPARIAN MICROCLIMATE

Characteristics of Forest Microclimates

Microclimate below forest canopies has been studied extensively for decades, though usually without explicit attention to riparian zones (FAO, 1962; Reifsnyder and Lull, 1965; Jarvis *et al.*, 1976; Rauner, 1976; Geiger *et al.*, 1995; McCaughey *et al.*, 1997; Chen *et al.*, 1999). Compared to open environments, the canopy reduces solar radiation, precipitation, and wind speed near ground level and increases longwave radiation received at the surface. These changes in turn influence the thermal and moisture environments under forest canopies.

Solar radiation transmission through forest canopies depends on the heights of the crown and the density and arrangement of foliage elements (Vézina and Petch, 1964; Reifsnyder and Lull, 1965; Federer, 1971; Black *et al.*, 1991). Reductions in solar radiation under forest cover range from more than 90 percent with dense canopies (Young and Mitchell, 1994; Chen *et al.*, 1995; Brosofske *et al.*, 1997; Davies-Colley *et al.*, 2000) to less than 75 percent in open stands (Örlander and Langvall, 1993; Spittlehouse *et al.*, 2004). The forest canopy changes the spectral distribution of light because plant foliage differentially absorbs and reflects the various wavelengths (Federer and Tanner, 1966; Vézina and Boulter, 1966; Atzet and Waring, 1970; Yang *et al.*, 1993). There is a greater reduction in the ultraviolet and photosynthetically active radiation ranges compared to longer solar radiation wavelengths. Longwave radiation to the forest floor increases as the canopy density increases because the forest canopy is usually warmer than the sky being blocked and has a higher emissivity (Reifsnyder and Lull, 1965). Although this increase somewhat offsets the reduction in solar radiation below the forest canopy, daytime net radiation below forest canopies is usually substantially lower than that in the open.

The amount of precipitation intercepted by the canopy and lost by evaporation depends upon tree species and the amount of canopy cover and typically varies from 10 to 30 percent of annual precipitation (Calder, 1990; McCaughey *et al.*, 1997; Pomeroy and Goodison, 1997; Spittlehouse, 1998). The fraction of precipitation intercepted decreases as storm magnitude and intensity increase. Time since the previous storm and weather conditions during the current storm are also important.

Wind speed under forest canopies is usually 10 to 20 percent of that in large openings (Raynor, 1971; Chen *et al.*, 1995; Davies-Colley *et al.*, 2000). Wind speed within forest openings depends on their size, and openings of less than about 0.1 ha will have low wind speeds, similar to those in the forest (Spittlehouse *et al.*, 2004).

Forest canopies tend to reduce the diurnal air temperature range compared to large open areas. Maximum differences (open area minus area under forest canopy) in daytime air temperature at the 1.5 to 2 m height varied from 3°C (Brosofske *et al.*, 1997; Davies-Colley *et al.*, 2000; Spittlehouse *et al.*, 2004) to 6°C or more (Young and Mitchell, 1994; Chen *et al.*, 1995; Cadenasso *et al.*, 1997). At night, air temperatures in forest areas are typically about 1°C higher than in the open (Chen *et al.*, 1995; Spittlehouse *et al.*, 2004), though Brosofske *et al.* (1997) found temperatures about 1°C lower above a stream. Surface and near-surface soil temperatures show the largest differences between forest and open sites, being up to 10 to 15°C lower under forest canopies during the daytime and

1 to 2°C higher at night (Chen *et al.*, 1995; Brosofske *et al.*, 1997; Spittlehouse *et al.*, 2004).

The vapor pressure of the air is mainly a function of the surrounding air mass and will be similar in the open and the forest. Consequently, the relative humidity and vapor pressure deficit will depend on the air temperature. The lower daytime forest air temperature means that relative humidity is typically 5 to 25 percent higher in the forest (Chen *et al.*, 1995; Brosofske *et al.*, 1997; Davies-Colley *et al.*, 2000; Spittlehouse *et al.*, 2004).

Riparian zones typically have elevated water tables and higher soil moisture than adjacent upland areas. Partly due to these hydrologic conditions, riparian forest cover and understory vegetation often differ from those of uplands, which would influence penetration of solar radiation and interception loss of precipitation. Surrounding slopes may also block direct and diffuse solar radiation. In small headwater streams, the riparian zone may be narrow to nonexistent due to topographic constraints imposed by steep side slopes (Richardson *et al.*, 2005). In addition to the effects of distinctive forest cover and higher soil moisture, riparian microclimate may be influenced by the stream channel, which can provide a local source of water vapor and act as a heat sink during the day, producing locally cooler and moister conditions near the stream (Brosofske *et al.*, 1997; Danehy and Kirpes, 2000). Riparian vegetation may also serve as a source of water vapor via transpiration (Danehy and Kirpes, 2000). Danehy and Kirpes (2000) found that enhanced relative humidity was restricted to a narrow zone within 10 m of the stream edge at 12 forested sites in eastern Oregon and Washington, most likely due to the constraining effects of steep local topography. Another topographic influence that is particularly important in mountain regions is the development of drainage winds that flow down valleys and gullies (Oke, 1987), advecting cool air into lower reaches.

Edge Effects and the Microclimate of Riparian Buffers

The magnitude of harvesting related changes in riparian microclimate will depend on the width of riparian buffers and how far edge effects extend into the buffer. Studies by Chen *et al.* (1993a,b, 1995) in an old-growth Douglas fir forest in Washington state (tree heights 50 to 65 m) are commonly cited in relation to edge effects and required buffer widths. Their results are consistent with those of Ledwith (1996), Brosofske *et al.* (1997), and Hagan and Whitman (2000), as well as with a range of other studies including Raynor (1971) (10.5 m tall red and white pine, closed canopy, New York state), Öerlander and Langvall (1993) (22 to 25 m tall Norway spruce and Scots pine stands of varying density, Sweden), Young and Mitchell (1994) (mixed podocarp-broadleaf forest in New Zealand), Cadenasso *et al.* (1997) (60+-year-old oak, birch, beech, and maple forest in New York state), Davies-Colley *et al.* (2000) (mature, 20 m tall native broadleaved rainforest in New Zealand), and Spittlehouse *et al.* (2004) (25 to 30 m tall Engelmann spruce-subalpine fir forest with a 40 percent canopy cover in British Columbia). All of these studies show that much of the change in microclimate takes place within about one tree height (15 to 60 m) of the edge. Solar radiation, wind speed, and soil temperature adjust to interior forest conditions more rapidly than do air temperature and relative humidity. Nighttime edge temperatures are similar to interior forest conditions. Daytime relative humidity decreases from interior to edge in response to the increased air temperature.

Edge orientation can be important, particularly for a south-facing edge (in the northern hemisphere), where solar radiation can penetrate some distance into the forest for much of the day. Dignan and Bren (2003) found that light penetration diminished rapidly within 10 to 30 m of the buffer edge for a riparian mountain ash forest in Australia, but that light penetration at 10 m was significantly greater for buffers that faced the equator than for other orientations. Wind blowing directly into the edge penetrates farther into the forest than from other directions (Raynor, 1971; Davies-Colley *et al.*, 2000).

Few studies appear to have examined microclimatic conditions within riparian buffers. In a study in northern California, above stream air temperatures measured in the early afternoon decreased with increasing buffer width, at decreases of about 1.6°C per 10 m for buffer widths up to 30 m and 0.2°C per 10 m for buffer widths from 30 m to 150 m (Ledwith, 1996). Above stream temperatures in the 150 m wide buffer treatments were about 6°C lower than at the no-buffer sites. In the same study, relative humidity was 10 to 15 percent higher than at a clear-cut site for 30 m wide buffers and increased another 5 to 10 percent as buffer widths increased to 150 m. At a study conducted at a first-order stream in Maine (Hagan and Whitman, 2000) where a 23 m wide buffer had been left on each side, air temperature 10 m from the stream in the buffer exhibited local differences from the reference sites of up to about 2°C. Differences up to about 4°C were observed within about 10 m from the buffer edge.

Only one study, covering 15 small streams in western Washington, appears to have examined changes in riparian microclimate using both pre-harvest and post-harvest data (Brosofske *et al.*, 1997). Prior to harvest, gradients from the stream into upland areas existed for all variables except solar radiation and wind speed. After harvest, conditions at the edges of riparian buffers tended to approximate those in the interior of the clear-cut. Solar radiation increased substantially within the buffers relative to pre-harvest conditions. Soil surface temperatures were higher after harvest. For buffers less than about 45 m wide (about one tree height), the pre-harvest gradient from riparian zone to upland was interrupted, which could influence habitat conditions for riparian fauna.

THERMAL PROCESSES AND HEADWATER STREAM TEMPERATURE

An understanding of thermal processes is required as a basis for understanding stream temperature dynamics, in particular for interpreting and generalizing from experimental studies of forestry influences. As a parcel of water flows through a stream reach, its temperature will change as a function of energy and water exchanges across the water surface and the streambed and banks (Figure 1) as described by the following equation (modified from Polehn and Kinsel, 2000).

$$\frac{dT_w}{dx} = \frac{\Sigma Q}{\rho C_p v D} + \frac{F_{gw}}{F}\left(T_{gw} - T_w\right) + \frac{F_{hyp}}{F}\left(T_{hyp} - T_w\right) \tag{1}$$

where dT_w/dx is the rate of change in the temperature (°C) of the water parcel with distance, x(m), as it flows downstream; ΣQ is the net heat exchange by radiation, turbulent exchange, and conduction across the water surface and bed (W/m^2); F is the streamflow (m^3/s); F_{gw} is the ground water inflow rate (m^3/s/m); F_{hyp} is the hyporheic exchange rate (m^3/s/m); T_{gw} and T_{hyp} are the ground water and hyporheic water temperatures, respectively (°C); ρ is the water density (kg/m^3); C_p is the specific heat of water (J/kg/°C); v is the local mean velocity (m/s); and D is the local mean depth (m). Equation (1) assumes steady state flow and ignores longitudinal dispersion. It also ignores the heat input of precipitation, which is typically much less than 1 percent of the total energy input to a stream (Webb and Zhang, 1997; Evans *et al.*, 1998). Similarly, frictional heating is neglected because it can be shown to be important relative to other energy exchanges only for steep streams with relatively high flows, under low radiation conditions. This section provides an overview of the dominant processes represented in Equation (1), followed by a discussion of spatial and temporal dynamics of stream temperature regimes.

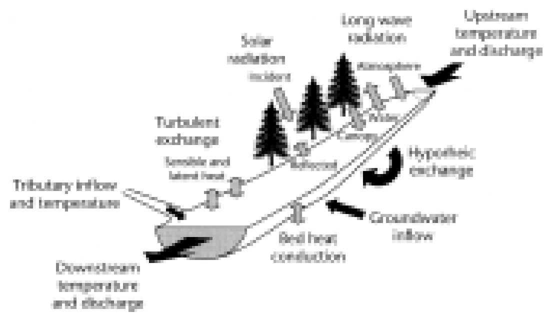

Figure 1. Factors Controlling Stream Temperature. Energy fluxes associated with water exchanges are shown as black arrows.

Radiative Exchanges

Radiation inputs to a stream surface include incoming solar radiation (direct and diffuse) and longwave radiation emitted by the atmosphere, forest canopy, and topography. Canopy cover along the sun's path will reduce the direct component of solar radiation, some of which will be scattered and transmitted through the canopy as diffuse radiation. Transmission of diffuse solar radiation will depend on both the spatial pattern of diffuse radiance from the sky dome and its interactions with the spatial arrangement of canopy elements. The details of solar radiation transmission through canopies are complex. It is often represented by simplified models based on extinction coefficients (e.g., Black *et al.*, 1991; Sridhar *et al.*, 2004) or the spatial distribution of canopy gaps (e.g., Dignan and Bren, 2003). Channel morphology can also influence incident solar radiation at a stream surface. Narrow, incised channels can be effectively shaded by streambanks (Pluhowski, 1972; Webb and Zhang, 1997). Wide channels tend to be less shaded because they have a canopy gap overhead, which will be particularly important for streams oriented north-south.

For solar elevation angles greater than 30 degrees, less than 10 percent of incoming solar radiation will be reflected from the water surface (Oke, 1987). Most incoming solar radiation thus enters the water column, where absorption can occur within the water column and at the bed (Evans *et al.*, 1998). The net effect is that roughly 90 to 95 percent of incident solar radiation is absorbed in the water column or at the bed and thus potentially available for stream heating.

except at low solar elevation angles (Evans *et al.*, 1998; Johnson, 2004).

Incoming longwave radiation will be a weighted sum of the emitted radiation from the atmosphere, surrounding terrain, and the canopy, with the weights being their respective view factors (Rutherford *et al.*, 1997). The water surface, canopy, and terrain have high emissivities (typically ≥ 0.95) (Oke, 1987), while the atmospheric emissivity is normally lower, except under overcast conditions. Outgoing longwave radiation includes that emitted by the water surface plus a small fraction (typically 3 to 8 percent) of the incoming longwave radiation that is reflected (Oke, 1987).

Peak daytime net radiation over a stream within a clear-cut can be more than five times greater than that under a forest canopy during summer (Brown, 1969), primarily due to the increase in incident solar radiation. Longwave radiation losses at night may be reduced slightly under forest canopy (Brown, 1969). It has been suggested that longwave radiation losses during autumn and winter may increase following removal (harvest) of forest canopy, leading to more rapid seasonal cooling (e.g., Macdonald *et al.*, 2003b), but this does not appear to have been investigated.

Sensible and Latent Heat Exchanges

Transfers of sensible and latent heat occur by conduction or diffusion and turbulent exchange in the overlying air. Sensible heat exchange depends on the temperature difference between the water surface and overlying air and on the wind speed. Where the stream is warmer than the air, heat transfer away from the stream would be promoted by the unstable temperature stratification, which enhances turbulence. Where the stream is cooler, heat transfer from the air to the stream would be dampened by the stable air temperature stratification (Oke, 1987). Evaporation and associated energy loss occur where the vapor pressure at the water surface (equal to the "saturation" value for the water temperature) exceeds the vapor pressure in the overlying air (a function of the air temperature and relative humidity); condensation and associated energy gain occur where the vapor pressure of the air exceeds the vapor pressure at the water surface. Latent heat exchange also depends on atmospheric stability over the stream.

Most field and modeling studies have used empirical "wind functions" to compute sensible and latent heat fluxes over small streams (e.g., Brown, 1969; Rutherford *et al.*, 1997; Webb and Zhang, 1997; Evans *et al.*, 1998; Johnson, 2004; Moore *et al.*, 2005). There can be great uncertainty in fluxes computed from wind functions, particularly because mean wind speeds under canopies may be less than the stall speed of typical anemometers (Story *et al.*, 2003).

Under intact forest cover, lack of ventilation appears to limit the absolute magnitude of sensible and latent heat exchanges over small streams (Brown, 1969; Webb and Zhang, 1997; Story *et al.*, 2003). Even at open sites such as clear-cuts, sensible and latent heat fluxes over small streams may be limited by bank sheltering, particularly for narrow, incised channels (Gulliver and Stefan, 1986). Brown (1969) and Moore *et al.* (2005) estimated the sensible and latent heat exchanges to be an order of magnitude lower than net radiation on sunny days in recent clear-cuts at coastal sites. Johnson (2004) computed higher values for latent heat flux at a stream in a recovering clear-cut in the Oregon Cascades, though it was still an order of magnitude lower than incident solar radiation.

Bed Heat Exchanges and Thermal Regime of the Streambed

Radiative energy absorbed at the streambed may be transferred to the water column by conduction and turbulent exchange and into the bed sediments directly by conduction and indirectly by advection (in locations where water infiltrates the bed). Given that turbulent exchange is more effective at transferring heat than conduction and that the flowing portions of streams are fully turbulent, much of the energy absorbed at the bed is transferred into the water column, and the temperature at the surface of the bed will generally be close to the temperature of the water column (Sinokrot and Stefan, 1993), except perhaps in pools with upwelling ground water or hyporheic exchange flow.

Bed heat conduction depends on the temperature gradients within the bed and its thermal conductivity and will normally act as a cooling influence on summer days and a warming influence at night, thus tending to reduce diurnal temperature range (Brown, 1985; Moore *et al.*, 2005). For streams within clear-cuts on sunny days, it has been estimated to be approximately 10 percent of net radiation in a step-pool stream (Moore *et al.*, 2005) and up to 25 percent in a bedrock channel (Brown, 1969). Bed heat conduction should depend on stream-subsurface interactions: stream reaches with upwelling ground water tend to have stronger daytime bed temperature gradients than those without and thus should have higher heat loss by conduction (Silliman and Booth, 1993; Story *et al.*, 2003).

Temperatures within the streambed are significant in their own right, since they may influence conditions for post-spawning egg development and fry emergence, as well as conditions for benthic invertebrates. Ringler and Hall (1975) observed summer bed temperature gradients in three catchments in the Oregon Coast Range. Gradients in an unlogged catchment were negligible. Differences of 2°C between the bed surface and 50 cm depth were observed in the streambed of a catchment subject to 25 percent patch-cut with riparian buffers, while bed temperatures in artificial redds in a fully clear-cut catchment reached 21°C with diurnal variations of up to 7°C at 25 cm depth and vertical changes of about 8°C over 50 cm. Bed temperatures varied greatly among locations within the clear-cut, likely due to variations in surface water exchange across the bed (Ringler and Hall, 1975). Consistent with this inference, Moore *et al.* (2005) found that bed temperatures in a step pool unit within a clear-cut followed stream temperature more closely in areas of downwelling flow into the bed than in areas of upwelling flow. Given the documented influence of subsurface hydrology on bed temperatures in a range of stream sizes and types and the potential interactions between stream temperature and stream subsurface exchanges (e.g., Shepherd *et al.*, 1986; White *et al.*, 1987; Silliman and Booth, 1993; Constantz, 1998; Curry *et al.*, 2002; Malcolm *et al.*, 2002; Alexander and Caissie, 2003; Moore *et al.*, 2005), the degree to which post-logging bed temperatures reflect changes in surface temperature likely depends on the local hydrologic environment.

Ground Water Inflow

Ground water is typically cooler than stream water in summer during daytime and warmer during winter and thus acts to moderate seasonal and diurnal stream temperature variations (Webb and Zhang, 1999; Bogan *et al.*, 2003). Forest harvesting can increase soil moisture and ground water levels due to decreased interception losses and transpiration (Hetherington, 1987; Adams *et al.*, 1991). Increases in ground water levels following forest harvesting could act to promote cooling or at least ameliorate warming. Alternatively, several authors have speculated that warming of shallow ground water in clear-cuts could result in heat advection to a stream, exacerbating the effects of increased solar radiation or decreasing the effectiveness of riparian buffers (e.g., Hewlett and Fortson, 1982; Hartman and Scrivener, 1990; Brosofske *et al.*, 1997; Bourque and Pomeroy, 2001), and this process has been incorporated into a catchment scale model of hydrology and water quality (St.-Hilaire *et al.*, 2000). Although there is ongoing research on the thermal response of ground water to forest harvesting (Alexander *et al.*, 2003), no published research appears to have examined ground water discharge and temperature both before and after harvest as a direct test of the ground water warming hypothesis.

Hyporheic Exchange

Hyporheic exchange is a two-way transfer of water between a stream and the saturated sediments in the bed and riparian zone. It often occurs where a stream meanders or where there are marked changes in stream gradient. For example, stream water typically flows into the bed at the top of a riffle and re-emerges at the bottom of the riffle (Harvey and Bencala, 1993). If the temperature of hyporheic water discharging into a stream differs from stream temperature, then hyporheic exchange can influence stream temperature dynamics (Equation 1). Several studies have shown that hyporheic exchange creates local thermal heterogeneity in larger streams (e.g., Bilby, 1984; Malard *et al.*, 2002), and recent studies suggest that it can be important in relation to both local and reach scale temperature patterns in headwater streams (Johnson, 2004; Moore *et al.*, 2005). However, there are significant methodological challenges associated with quantifying rates of hyporheic exchange and its influence on stream temperature (Kasahara and Wondzell, 2003; Story *et al.*, 2003; Moore *et al.*, 2005).

Tributary Inflow

Effects of tributary inflow depend on the temperature difference between inflow and stream temperatures and on the relative contribution to discharge, according to a simple mixing equation.

$$T_m = f_i T_i + (1 - f_i) T_s = T_s + f_i (T_i - T_s) \quad (2)$$

where T_i is the inflow temperature (°C); T_s is temperature at the upstream end of the reach (°C); T_m is the temperature of the stream inflow mixture (°C); and f_i is the ratio of inflow rate to streamflow at the downstream end of the reach. Equation (2) assumes complete mixing and may not be valid in the immediate vicinity and some distance downstream of the tributary mouth, where lateral mixing of the tributary flow with the main stream may be incomplete.

Longitudinal Dispersion and Effects of Pools

Longitudinal dispersion results from the variation in velocity through the cross-section of a stream. It would act to "smooth" temperature waves as they propagate downstream, potentially causing a progressive decrease in the diurnal temperature maximum as clearing heated water flows downstream through forested reaches. It is often assumed to be negligible in modeling studies of both small and large streams (e.g., Sinokrot and Stefan, 1993; Rutherford *et al.*, 1997; Polehn and Kinsel, 2000), but no published studies appear to have evaluated its influence in small streams.

The presence of pools can also potentially influence stream temperatures. Being locally deeper zones, pools would tend to change temperature more slowly than the shallower, flowing portions of the stream. However, Brown (1972) observed that there was incomplete mixing in many pools in pool riffle streams in Oregon such that the effective width and depth of flowing water through pools were much smaller than the pool dimensions. Thermal influences of pools do not appear to have been examined in smaller, steeper step pool streams.

Equilibrium Temperature and Adjustment to Changes in Thermal Environment

For a given set of boundary conditions (e.g., solar radiation, air temperature, humidity, wind speed), there will be an "equilibrium" water temperature that will produce a net energy exchange of zero and thus no further change in temperature as water flows downstream (i.e., $dT_w/dx = 0$; Edinger *et al.*, 1968). For stream water being warmed as it flows through a clear-cut, the equilibrium temperature represents the maximum possible temperature the parcel could achieve within the reach at a given time, assuming that boundary conditions remain constant in time and space. However, equilibrium temperature may not be achieved because the boundary conditions may change in time or space before the water parcel can adjust fully to the thermal environment. The concept applies most simply to streams or time scales for which the energy exchanges across the air/water interface dominate the energy budget (Edinger *et al.*, 1968). Stream temperatures influenced by substantial ground water inputs will be consistently less than equilibrium temperature computed from atmospheric conditions during summer and higher in winter (Bogan *et al.*, 2003). Equilibrium temperatures for unshaded reaches are higher than those under shade during summer afternoons (Bartholow, 2000; Bogan *et al.*, 2003).

The rate at which a parcel of water adjusts to a change in the thermal environment depends on stream depth because for deeper streams, heat would be added to or drawn from a greater volume of water. Shallow streams should thus adjust relatively quickly to a change in thermal environment. In addition, flow velocity influences the length of time the parcel of water is exposed to energy exchanges across the water surface and the bed and thus the extent to which the parcel can adjust fully to its thermal environment within a given reach (Figure 2). Given that the depth and velocity of a stream tend to increase with discharge, the sensitivity of stream temperature to a given set of energy inputs should increase as discharge decreases (Brown, 1985; Beschta *et al.*, 1987; Moore *et al.*, 2005).

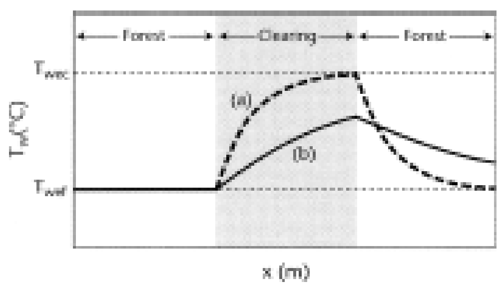

Figure 2. Schematic Temperature Patterns Along a Stream Flowing From Intact Forest, Through a Clear-Cut, and Back Under Intact Forest for (a) Shallow, Low Velocity, and (b) Deep, High Velocity Conditions (T_{wef} = equilibrium temperature in forest; T_{wec} = equilibrium temperature in clearing).

Thermal Trends and Heterogeneity Within Stream Networks

Small forest streams tend to be colder and exhibit less diurnal variability than larger downstream reaches, up to about fourth or fifth order (Vannote and Sweeney, 1980; Holtby and Newcombe, 1982; Macdonald *et al.*, 2003a). Small streams will be more heavily shaded by riparian vegetation and near stream terrain, will have a higher ratio of ground water inflow in a reach to the total downstream flow, and are located at higher elevations and thus experience a generally cooler thermal environment. However, local deviations from a dominant downstream warming trend may occur as a result of ground water inflow, hyporheic exchange, or thermal contrasts between isolated pools and the flowing portion of a stream. In addition, lakes, ponds, and wetlands can produce elevated water temperatures at their outlets, resulting in downstream cooling below them over distances of hundreds of meters, even through cut blocks (Mellina *et al.*, 2002).

Thermal heterogeneity at a range of spatial scales has been well documented in intermediate and large streams (i.e., third order and larger; Bilby, 1984; Arscott *et al.*, 2001; Malard *et al.*, 2001; Ebersole *et al.*, 2003), where it is an important aspect of stream habitat (Neilsen *et al.*, 1994; Ebersole *et al.*, 2003). Thermal heterogeneity in small streams has apparently received less attention, though Story *et al.* (2003) and Moore *et al.* (2005) observed substantial temperature variations in small streams for reaches within a clear-cut and downstream of forest clearings, both along the reach and within channel units.

Stratification of pools can be an ecologically important source of thermal heterogeneity, although its occurrence is variable. Brown (1972) found that only one pool in an intermediate-sized stream with a pool-riffle morphology exhibited significant vertical stratification, with a temperature decrease of 6.5°C over 1.2 m depth. Nielsen *et al.* (1994) observed more prevalent thermal stratification in pools in three larger rivers in northern California and noted their significance as thermal refugia for steelhead. No published studies appear to have examined stratification of pools in smaller, steeper streams.

STREAM TEMPERATURE RESPONSE TO FOREST MANAGEMENT

The effects of forest management on stream temperature have been estimated using a variety of study designs. The most rigorous approach is the BACI (before-after/control-impact) design, which involves monitoring both before and after treatment and includes untreated control sites (e.g., Harris, 1977). A variation is to use a regression of stream temperature on weather data in place of a calibration with a control catchment (e.g., Holtby and Newcombe, 1982; Curry *et al.*, 2002). Some studies used synoptic surveys of streams that had been subjected to a range of treatments (e.g., Rashin and Graber, 1992; Mellina *et al.*, 2002), while others monitored downstream temperature changes in clear-cuts (Brownlee *et al.*, 1988). This review focuses primarily on studies employing a BACI design, which are summarized in Table 1.

Influences of Forest Harvesting Without Riparian Buffers

Almost all study streams in rain-dominated catchments experienced post-harvest increases in summer temperatures, with increases in summer maximum temperatures ranging up to 13°C (Table 1). The strong response at Needle Branch may reflect the harsh treatment: clear-cutting to the streambank, slash burning, and removal of wood from the stream. The difference in response between Needle Branch and H.J. Andrews (HJA) Watershed 1, which was subjected to similar treatment, may reflect the differences in aspects (i.e., south for Needle Branch versus northwest for HJA Watershed 1), but other factors also could have influenced the responses. At HJA Watershed 3, where streamside harvesting influenced only part of the stream length, a debris torrent removed riparian vegetation and scoured the channel to bedrock, ultimately leading to similar temperature increases as observed in HJA Watershed 1. At HJA Watersheds 1 and 3, the timing of summer maximum temperatures shifted from August for predisturbance conditions into late June and early July after disturbance, probably because inputs of solar radiation came to dominate other factors such as seasonal variations in discharge (Johnson and Jones, 2000).

In contrast to the results summarized in Table 1, Jackson *et al.* (2001) found that daily maximum temperature for four of seven study streams within clear-cuts in the Washington Coast Range either did not change significantly or decreased following harvesting, likely due to the large volumes of slash that covered the streams and provided shade. However, the post-harvest summer was substantially cooler than the pre-harvest summer, possibly confounding the results.

Effects on summer minimum daily temperatures do not appear to be as marked as those on maximum temperatures, with both small increases and decreases (on the order of 1 to 2°C) having been reported (e.g., Feller, 1981; Johnson and Jones, 2000). Summer daily temperature ranges after logging have increased up to about 7 to 8°C, compared to pre-logging ranges of about 1 to 3°C (Feller, 1981; Johnson and Jones, 2000). Carnation Creek and one of its tributaries experienced smaller increases in diurnal temperature range than found in other studies, but the reason is not obvious from available information (Holtby and Newcombe, 1982).

Fewer studies have examined stream temperature response to forest harvesting in snowmelt-dominated regimes, and no published studies employed a BACI design to estimate effects of no-buffer harvesting in these environments. Brownlee *et al.* (1988) measured downstream increases in summertime mean daily temperature of 1 to 3°C in three small streams flowing through clear-cuts in the central interior of British Columbia (BC), with increases in daily maximum temperatures of 4.5 to 9°C on the warmest days. Assuming that downstream temperature changes in these reaches were modest under pre-logging conditions, these upstream/downstream comparisons

TABLE 1. Summary of Experiments Documenting Stream Temperature Changes after Forest Harvesting.

Study Location	Latitude (°N)	Treatment Catchment	Harvesting Type[1]	Riparian Buffer	Aspect	Temperature Variable	Observed Value After Treatment (°C)	Change Due to Treatment (observed-predicted) (°C)	Recovery to Pre-Treatment Conditions	Reference
RAIN DOMINATED										
Oregon Coast Range (Alsea Watershed)	45	Needle Branch Creek (71 ha)	CC (100%)	no buffer	S	Mean of monthly max. T (Apr.-Oct.)	17.8	8.8	~70% recovery in 7 years	Harris, 1977
						Maximum summer T	26	11.6	~70% recovery in 7 years	
Oregon Coast Range (Alsea Watershed)	45	Deer Creek (304 ha)	PC (25%)	30 m	S	Mean of monthly max. T (Apr.-Oct.)	14.8	2.0	No obvious recovery over 7 years	Harris, 1977
British Columbia, Southern Coast Mountains	49	A[2] (59 ha)	CC (20%)	no buffer	SSW	Maximum difference between observed and predicted for daily max. T		5.0	No obvious recovery in 4 years	Moore et al., 2005
British Columbia, Southern Coast Mountains	49	A[2] (23.1 ha)	CC (61%)	no buffer	S	Maximum recorded T	21.8	3.9[3]	Apparently full recovery after 6-7 years	Feller, 1981
British Columbia, Southern Coast Mountains	49	B (68 ha)	CC (19%) followed by slash burn	no buffer	S	Maximum recorded T	20.3	1.8[3]	No apparent recovery after 7 years	Feller, 1981
Oregon Cascades (H.J. Andrews)	45	WS1 (96 ha)	CC (100%)	no buffer	WNW	Maximum summer T	23.9	~7[4]	Apparently full recovery in 15 years	Johnson and Jones, 2000
						Summer mean weekly max. T	not given	5.4 to 6.4 (first 4 years after logging)[4]		
						Summer mean weekly min. T	not given	1.8 to 2.0 (higher)[4]		
Oregon Cascades (H.J. Andrews)	45	WS3 (101 ha)	PC (25%)	Riparian vegetation removed by debris flow after logging	NW	Maximum summer T	23.9	~7[4]	Apparently full recovery in 15 years	Johnson and Jones, 2000

TABLE 1. Summary of Experiments Documenting Stream Temperature Changes after Forest Harvesting (cont'd.).

Study Location	Latitude (°N)	Treatment Catchment	Harvesting Type[3]	Riparian Buffer	Aspect	Temperature Variable	Observed Value After Treatment (°C)	Change Due to Treatment (observed-predicted) (°C)	Recovery to Pre-Treatment Conditions	Reference
RAIN DOMINATED (cont'd.)										
Oregon Cascades (H.J. Andrews) (cont'd.)						Summer mean weekly max. T	not given	3.5 to 5.2 (first 3 years after disturbance)[4]		
						Summer mean weekly min. T	not given	-0.1 to 1.0 (first 3 years after disturbance)[4]		
Oregon Cascades (Bull Run)	45	FC1 (89 ha)	PC (25%) and burned	"sparse strips" left on south banks	W	Maximum summer T	15	3.0	Effect on max. T decreased to < 1°C within 6 years	Harr and Fredriksen, 1988
Oregon Cascades (Bull Run)	45	FC3 (71 ha)	PC (25%)	"sparse strips" left on south banks	SW	Maximum summer T	16	2.5	Effect on max. T decreased to < 1°C within 6 years	Harr and Fredriksen, 1988
Vancouver Island, British Columbia (Carnation Creek)	49	J tributary (24 ha)	CC (100%)	no buffer		Summer (JJA) diel T range	2.1	1.6[5] (after logging)	Only one post-treatment year	Holtby and Newcombe, 1982
							3.2	2.8[5] (after logging and burning)	Only one post-treatment year	
Vancouver Island, British Columbia (Carnation Creek)	49	H tributary (12 ha)	CC (100%)	no buffer		Summer (JJA) diel T range	1.8	1.4[5]	Only one post-treatment year	Holtby and Newcombe, 1982
INTERIOR										
Central Interior of BC (Stuart-Takla FFIP)	55	B8 (42.5 ha)	CC (18%)	10-20 m, all trees > 20 cm dbh harvested	NW	Weekly T_{mean} (max. change)	not given	2.5	No apparent recovery over 5 years	Macdonald *et al.*, 2003b
Central Interior of BC (Stuart-Takla FFIP)	55	B5 (180 ha)	CC (40%)	10-20 m, all trees > 15-20 cm dbh harvested	NW	Weekly T_{mean} (max. change)	not given	3.0	No apparent recovery over 5 years	Macdonald *et al.*, 2003b

TABLE 1. Summary of Experiments Documenting Stream Temperature Changes after Forest Harvesting (cont'd.).

Study Location	Latitude (°N)	Treatment Catchment	Harvesting Type[1]	Riparian Buffer	Aspect	Temperature Variable	Observed Value After Treatment (°C)	Change Due to Treatment (observed-predicted) (°C)	Recovery to Pre-Treatment Conditions	Reference
INTERIOR (cont'd.)										
Central Interior of BC (Stuart-Takla FFIP)	55	B2 (18 ha)	CC (89%)	20 m high retention buffer on lower 60% of stream length within cut block	W	Weekly T_{mean} (max. change)	not given	3.8	No apparent recovery over 5 years	Macdonald *et al.*, 2003
Central Interior of BC (Stuart-Takla FFIP)	55	B1 (313 ha)	CC (6%)	20 m high retention	W	Weekly T_{mean} (max. change)	not given	0.8	No apparent recovery over 5 years	Macdonald *et al.*, 2003b
Central Interior of BC (Stuart-Takla FFIP)	55	C6 (26 ha)	CC (90%)	20 m low retention	NE	Weekly T_{mean} (max. change)	not given	At least 5.4 (missing data)	No apparent recovery over 5 years	Macdonald *et al.*, 2003b
Central Interior of BC (Stuart-Takla FFIP)	55	118-48 (410 ha)	CC (13%)	30 m, all commercial trees harvested	SW	Mean T_{max} in Aug.		0.5	No apparent recovery over 3 years	Mellina *et al.*, 2002
						Mean T_{min} in Aug.		-0.2	No apparent recovery over 3 years	
						T_{max} in Aug.	20.1	2.2	Insufficient info.	
Central Interior of BC (Stuart-Takla FFIP)	55	118-16 (310 ha)	CC (9%)	30 m, all commercial trees harvested	SE	Mean T_{max} in Aug.		0.3	No apparent recovery over 3 years	Mellina *et al.*, 2002
						Mean T_{min} in Aug.		-1.1	No apparent recovery over 3 years	
						T_{max} in Aug.	20.1	5.1	Insufficient info.	

[1]CC = clear-cut, PC = patch cut and number in brackets is % of catchment area treated.
[2]Different creeks with same name.
[3]Computed as difference in maximum observed temperatures between treatment and control streams after logging, compared to difference before logging.
[4]Computed by authors as difference between treatment and control streams due to lack of pre-logging regression.
[5]Computed as difference pre-logging and post-logging for the treatment stream due to lack of calibration with control.

provide an estimate of the effect of clear-cut logging. Winkler *et al.* (2003) inferred similar effect sizes by comparing summer water temperatures for small, high-elevation streams in the southern interior of BC, one in a clear-cut and one in undisturbed forest.

Winter temperatures have received less attention. Feller (1981) found short lived, modest increases in winter temperatures following logging and decreases following logging and slash burning, though there was no clear explanation for these divergent patterns. Post-harvest temperature differences between clear-cut Needle Branch and Flynn Creek (the control) were positive during winter, though smaller than summer differences (Brown and Krygier, 1970). In rain dominated catchments, smaller effects would be expected in winter than in summer, based on the lower energy inputs and higher discharges. In small snowmelt fed catchments, particularly at high elevation or northern sites, ice formation and snow cover within the channel should reduce temperatures to near 0°C regardless of canopy cover (e.g., Mellina *et al.*, 2002; Macdonald *et al.*, 2003b), except possibly in ground water discharge areas.

Influences of Harvesting With Riparian Buffers

Studies in rain dominated catchments suggest that buffers may reduce but not entirely protect against increases in summer stream temperature. In the Oregon Coast Range, the mean of the summer monthly maximum temperatures increased by only 2°C at buffered Deer Creek, compared to the 5.5°C increase observed at unbuffered Needle Branch (Harris, 1977; Table 1). However, this comparison is confounded by the fact that the Deer Creek watershed was 25 percent patch-cut, with only a portion of the stream network adjacent to cut blocks, compared to the 100 percent cutting at Needle Branch. Post-logging increases in maximum summer stream temperature of up to 3°C were observed at the two Fox Creek streams in the Oregon Cascades, where sparse or partial-retention buffers were left (Harr and Fredriksen, 1988). In the Washington Coast Range, post-harvest changes in daily maximum temperature ranged from -0.5°C to 2.6°C for three streams with unthinned buffers (15 to 21 m wide), while streams with buffers of nonmerchantable species warmed by 2.8 to 4.9°C (Jackson *et al.*, 2001).

Two studies in snowmelt dominated subboreal catchments examined stream temperature response to harvesting with partial retention buffers, both conducted as part of the Stuart-Takla Fish-Forestry Interaction Project in the central interior of BC (Mellina *et al.*, 2002; Macdonald *et al.*, 2003b). Macdonald *et al.* (2003b) reported maximum changes in mean weekly temperatures that ranged from less than 1°C to more than 5°C for a set of streams subject to a range of forestry treatments (Table 1). Greater warming was observed for the low retention buffers and a patch retention treatment than for the high retention buffers. The protective effect of the buffers was compromised by significant blowdown, which reduced riparian canopy density from about 35 percent to 10 percent at one high retention buffer and from about 15 percent to less than 5 percent at one low retention buffer. Mellina *et al.* (2002) documented temperature responses to clear-cut logging with riparian buffers for two lake headed streams. Both streams cooled in the downstream direction both before and after logging. Mean August temperatures at the downstream ends of the cut blocks were slightly warmer (less than 1°C) after logging, although the maximum daily temperature in August increased by more than 5°C at one stream. The dominant downstream cooling observed both before and after harvest was attributed to the combination of warm source temperatures associated with the lakes and the strong cooling effect of ground water inflow through the clear-cut, as well as the residual shade provided by the partially logged riparian buffer.

Thermal Recovery Through Time

Post-harvest summer stream temperatures should decrease through time as riparian vegetation and shade levels recover. Summers (unpublished, cited in Beschta *et al.*, 1987) found that shade levels at sites that had been clear-cut and burned recovered more rapidly in wetter forest types and at lower elevations. Shade recovery to old-growth levels occurred within about 10 years in the Coast Range western hemlock zone and about 20 years in the Cascade Mountain western hemlock zone. Shade recovery was only 50 percent complete after about 20 years in the higher-elevation Pacific silver fir zone in the Cascades. Shade recovery depends not only on vegetation growth but also stream width: narrow streams should recover more rapidly.

In experimental studies, temperature recovery occurred within 5 to 10 years or was at least under way for several rain dominated streams (Brown and Krygier, 1970; Harris, 1977; Feller, 1981; Harr and Fredriksen, 1988). However, recovery took longer in other cases or was not detectable in the post-harvest period in some cases. Johnson and Jones (2000) found that summer stream temperatures recovered after about 15 years for streams that had their channels and riparian zones disturbed by debris flows in the Oregon Cascades, while Feller (1981) found no evidence of recovery seven years after harvest for a catchment subject to logging and slash burning. In the subboreal environment of B.C., Mellina *et al.* (2002) found no evidence of recovery within the first three years, while Macdonald *et al.* (2003b) found no evidence for recovery of summer temperatures within the first five years following harvesting with partial-retention buffers. Because the streams studied by Macdonald *et al.* (2003b) were well shaded by shrubby vegetation both before and after harvest (E. MacIsaac, Fisheries and Oceans Canada, November 29, 2004, personal communication), it appears that shading by low vegetation may not be as effective at maintaining low stream temperatures as that from trees. In addition, blowdown within the buffers may have contributed to the apparent lack of recovery reported by Macdonald *et al.* (2003b).

Comparison With Studies Outside the Pacific Northwest

Studies of the effects of forestry on stream temperature have been conducted at locations outside the PNW, including Great Britain (Stott and Marks, 2000), eastern and southern United States (e.g., Swift and Messer, 1971; Hewlett and Fortson, 1982; Rishel *et al.*, 1982; Lynch *et al.*, 1984), Quebec (Prevost *et al.*, 1999), and New Zealand (Rowe and Taylor, 1994). Consistent with results from the PNW, these studies have found that streams subject to canopy removal become warmer in the summer and exhibit greater diurnal fluctuations. However, differences in environmental conditions (climate, hydrology, vegetation), forestry treatments, and reported temperature metrics limit the comparability of quantitative results.

Effects of Forest Roads

Forest roads and their rights-of-way would have a similar influence to cut blocks in terms of enhanced solar radiation inputs. Brown *et al.* (1971) observed downstream warming of up to 7°C in a 46 m reach of Deep Cut Creek in Oregon, which was completely cleared of vegetation during road construction. In the central interior of B.C., streams warmed over 2°C across a 50 m right-of-way, 1.4°C across a 30 m right-of-way, and about 0.4°C across a 20 m right-of-way (Herunter *et al.*, 2003). Another possible effect of forest roads is the interception of ground water and its conveyance to a stream via ditches, where it is exposed to solar radiation, effectively replacing the cooling effect of ground water inflow with inflow of warm ditch water. This process has been observed in the central interior of B.C. (D. Maloney, B.C. Ministry of Forests, Northern Interior Region, October 3, 2000, personal communication) and may be most important in low relief terrain, where high water tables could maintain ditch flow during periods of warm weather.

Downstream and Cumulative Effects

The potential for cumulative effects associated with warming of headwater streams is a significant management concern. Beschta and Taylor (1988) demonstrated that forest harvesting between 1955 and 1984 in the 325 km^2 Salmon Creek watershed produced substantial increases in summer water temperature at the mouth of the watershed. Given that current forest practices in the Pacific Northwest require or recommend buffers around all but the smallest streams and require more careful treatment of unstable terrain, cumulative effects resulting from current practices may be of lower magnitude than those found by Beschta and Taylor (1988). At smaller scales, downstream transmission of clearing heated water would increase the spatial extent of thermal impacts and possibly reduce the habitat value of localized cool water areas that form where headwater streams flow into larger, warmer streams, which tend to be cooler and have higher dissolved oxygen concentrations than other types of cool water areas (Bilby, 1984).

Some authors have argued that downstream cooling is unlikely to occur except in association with cooler ground water or tributary inflow (e.g., Beschta *et al.*, 1987), while others have contended that streams can recover their natural thermal regimes within relatively short distances downstream of forest openings (e.g., Zwieniecki and Newton, 1999). Streams can cool in the downstream direction by dissipation of heat out of the water column or via dilution by cool inflows. Dissipation to the atmosphere (and thus out of the stream-riparian system) can occur via sensible and latent heat exchange and longwave radiation from the water surface. Heat loss via evaporation (latent heat) can be a particularly effective dissipation mechanism at higher water temperatures for larger streams (Benner and Beschta, 2000; Mohseni *et al.*, 2002). However, the effectiveness of evaporation may be reduced in small forest streams by negative feedback caused by accumulation of water vapor above the stream due to poor ventilation. Dissipation of heat from the water column into the bed can occur via conduction and hyporheic exchange (assuming the bed and hyporheic zone are cooler than stream water), but reciprocally, these mechanisms would add that heat to the bed and hyporheic zone (Poole *et al.*, 2001). Therefore, cooling of the water column may occur at the expense of warming the streambed and riparian zone, which can influence rates of growth and development of benthic invertebrates and influence salmonid incubation (Vannote and Sweeney, 1980; Crisp, 1990; Malcolm *et al.*, 2002).

Reported downstream temperature changes below forest clearings are highly variable, with some streams cooling but others continuing to warm (e.g., McGurk, 1989; Caldwell *et al.*, 1991; Zwieniecki and Newton, 1999; Story *et al.*, 2003). The maximum cooling reported in the literature was almost 7°C over a distance of about 120 m (Greene, 1950). The magnitude of downstream cooling may be positively related in some cases to the maximum upstream temperature. Keith *et al.* (1998) found that greater cooling occurred on sunny days, when maximum stream temperatures were greater than 20°C, than on cloudy days, when maximum stream temperatures were only approximately 13°C. Storey and Cowley (1997) observed downstream cooling of 1 to 2°C for two streams in New Zealand where upstream temperatures were 20°C or greater. In a third stream, which had a narrow margin of forest in the riparian zone upstream of the study reach, upstream temperatures were lower, approximately 17°C, and no downstream cooling was observed. However, a high upstream temperature does not ensure that downstream cooling will occur, as illustrated by Brown *et al.* (1971), who observed no significant cooling despite an upstream temperature of 29°C. These studies all employed only post-treatment data, so that even where cooling was observed, there is no basis to assess whether the stream temperature had recovered to pre-logging levels.

Of the studies reviewed, only three attempted to quantify the processes governing downstream temperature changes under shade (Brown *et al.*, 1971; Story *et al.*, 2003; Johnson, 2004). For one clear July day, Brown *et al.* (1971) found that the latent and conductive heat fluxes were the only cooling (negative) terms because ground water inflow was negligible, and these were offset by the warming influences of net radiation and sensible heat, even though the forest canopy substantially reduced inputs of solar radiation. This estimated net input of heat is consistent with the observed lack of significant downstream cooling. Story *et al.* (2003) found that radiative and turbulent energy exchanges at heavily shaded sites on two streams represented a net input of heat during most afternoons and therefore could not explain the observed cooling of up to more than 4°C over distances of less than 150 m. Instead, downstream decreases in daily maximum temperatures were caused by energy exchanges between the streams and their subsurface environments via ground water inflow, hyporheic exchange, and heat conduction. In contrast, Johnson (2004) demonstrated that downstream cooling could occur in an artificially shaded stream with no ground water inflow or hyporheic exchange. Clearly, more research is required to clarify the mechanisms responsible for downstream cooling and how they respond to local conditions.

Three factors may mitigate against cumulative effects of stream warming. First, although cooling by dilution of streamwater with colder inflow water cannot reduce downstream temperatures to pre-harvest levels, dilution may be great enough, especially at larger spatial scales, to render the changes ecologically insignificant, as long as the total discharge of clearing-heated streams is not a substantial fraction of the total discharge (Equation 2). Second, the effects of energy inputs will not be linearly additive throughout a stream network. This is a consequence of the relation between energy exchange (particularly energy losses via evaporation and longwave radiation) and stream temperature: increased temperatures in one reach due to reduction of riparian shade may reduce the propensity for the stream to warm in downstream reaches, even in the absence of dilution by ground water or tributary inflow. Finally, where streams flow into lakes, ponds, or wetlands, the resetting of stream temperatures may minimize the possibility for cumulative effects below the lentic environment (Ward and Stanford, 1983).

An important aspect of cumulative effects is the indirect impacts of forest harvesting. For example, removing riparian vegetation not only reduces shade but can result in a stream becoming wider and shallower due to bank erosion, which can produce a greater temperature response to the additional heat inputs. Aggradation caused by logging related mass movements and subsequent sediment loading can similarly cause stream widening and promote warming (Beschta and Taylor, 1988). In addition, debris flows that remove vegetation and scour channel beds to bedrock can lead to marked warming in headwater tributaries (Johnson and Jones, 2000).

MONITORING AND PREDICTING STREAM TEMPERATURE AND ITS CAUSAL FACTORS

Successful management of forestry operations for maintenance of stream temperature regimes requires accurate, cost effective tools for monitoring stream temperature and its causal factors and for predicting the effects of different harvesting options.

Monitoring Stream Temperature

Most recent studies have employed submersible temperature loggers to monitor temperature. These are relatively inexpensive and sufficiently accurate (typically within 0.2°C) for forestry related applications. They also provide sufficient temporal resolution to allow calculation of temperature metrics at a range of time scales, such as maximum daily temperature and accumulated seasonal degree days. Multiple loggers should be used within and downstream of clearings to avoid sampling problems resulting from small scale spatial variability (Story *et al.*, 2003; Moore *et al.*, 2005).

Forward looking infrared radiometry from helicopters has been used for investigating stream temperature patterns in medium to large streams (Torgerson *et al.*, 1999, 2001). However, its application to headwater streams is limited by the sensor resolution relative to typical channel widths for small streams and the fact that low vegetation overhanging the channel may obscure the water surface. However, the technology may be invaluable in identifying cool water areas at tributary mouths and their significance as thermal refugia.

Measuring Shade

Given the importance of solar radiation in causing stream warming following forest harvesting, reliable and practical methods for measuring shade are required for use as indicators of the effectiveness of riparian buffers in protecting against stream temperature changes and for use in predictive models of stream temperature. Many models use canopy and terrain angles, either field measured with a clinometer or estimated from the geometry of the riparian canopy and stream, to determine whether direct solar radiation is blocked. Where blockage by vegetation occurs, the direct radiation reaching the stream is reduced according to estimates of the transmissivity or shade density of the riparian canopy (e.g., Beschta and Weatherred, 1984; Rutherford *et al.*, 1997; Sridhar *et al.*, 2004).

Ocular estimates of canopy cover using instruments such as a spherical densiometer are often used as indices or as model input (e.g., Sullivan *et al.*, 1990; Mellina *et al.*, 2002). Although ocular instruments are generally inexpensive and easy to use in the field, they are prone to operator error due to subjective interpretation. In addition, measurements such as spherical density may not provide a good index of solar radiation blockage except in a uniform canopy. Brazier and Brown (1973) developed an instrument for measuring angular canopy density (ACD), which is the canopy density in the portion of the sky through which the sun passes during the time of maximum potential stream heating, typically July or August, depending on location and hydrologic regime. Teti (2001) described an alternative, robust instrument for measuring ACD based on a convex mirror. Another instrument, the Solar Pathfinder™, focuses on the portion of the canopy responsible for blocking direct solar radiation throughout the day.

Hemispherical photography offers an alternative that is less prone to operator error than ocular methods and allows computation of a range of parameters that are strongly related to solar radiation exposure (Ringold *et al.*, 2003), but it requires off-site analysis. Digital cameras that can be used with fish-eye lenses are steadily decreasing in price, and functional software packages are available both commercially and by free distribution (Frazer *et al.*, 1999).

Shade can also be characterized by comparing radiation or light levels measured above the stream to those at an open site. For example, Webb and Zhang (1997) used a hand-held photographic light meter, following Barthelow (1989), while Davies-Colley and Payne (1998) used a leaf area index canopy analyzer.

Although studies have compared canopy density parameters estimated by different methods (e.g., Englund *et al.*, 2000; Ringold *et al.*, 2003), few studies appear to have assessed which approach provides the best measure of shade for stream temperature assessment. Brazier and Brown (1973) estimated the amount of "heat blockage" caused by the canopy cover in riparian buffers by comparing observed water temperatures to temperatures estimated for a situation of no canopy shade. The good relation between estimated heat blockage and measured ACD confirmed the relevance of ACD as an indicator of buffer effectiveness for temperature control. Rutherford *et al.* (1997) found substantial sampling variability in their shade estimates for a small stream in New Zealand. Using the average field measured shade value in the physically based model STREAMLINE resulted in overestimates of stream temperature. Moore *et al.* (2005) used the spatial distribution of canopy gaps derived from hemispherical canopy photographs, in conjunction with measurements of total and direct solar radiation at an open site, to model the temporal variation of solar irradiance at a stream surface for a clear sky day. Their inability to close a reach scale energy budget may have resulted from sampling bias associated with the canopy photographs but could also have arisen from errors in estimates of the other energy exchanges. Further work is needed to verify predicted solar radiation based on shade measurements, ideally using solar radiation measurements to avoid confounding factors involved in stream heat budgets.

These efforts will be particularly important for application in complex shade environments such as partial-retention riparian buffers or variable retention harvesting units.

In addition to the quantitative measurement of shade, there are questions about shade "quality" in terms of minimizing energy inputs to a stream. For example, Hewlett and Fortson (1982) presented evidence that shade from low, brushy vegetation was less effective than taller trees at moderating water temperatures for a stream in the Georgia Piedmont. Similarly, Macdonald *et al.* (2003b) observed significant temperature increases in central BC despite cover by low vegetation. If these effects are real, it may be that overhanging low vegetation transmits more solar radiation than a coniferous canopy that obstructs the same fraction of sky view, or that it promotes net energy inputs to a stream by influencing longwave radiation and sensible and/or latent heat.

Predicting the Influences of Forest Harvesting on Stream Temperature

Empirical models for predicting stream temperature response to forest harvesting in the PNW include Mitchell's (1999) regression model for predicting the mean monthly stream temperature following complete removal of the riparian canopy, a "temperature screen" for predicting stream temperature as a function of elevation and percent stream shade in Washington (Sullivan *et al.*, 1990) and a multiple regression model that predicts downstream temperature changes as a function of upstream temperature and canopy cover in the central interior of B.C. (Mellina *et al.*, 2002). Although empirical models have the virtues of simplicity and low requirements for input data, they usually involve significant uncertainties, especially when applied to situations different from those represented in the calibration data (e.g., different locations, weather conditions).

Physically based models incorporating energy balance concepts have been developed for application to individual stream reaches, including the seminal model introduced by Brown (1969, 1985), TEMP-84 (Beschta and Weatherred, 1984), TEMPEST (Adams and Sullivan, 1989), Heat Source (Boyd, 1996), and STREAMLINE (Rutherford *et al.*, 1997). Models to simulate stream temperatures at the stream network or catchment scale include SNTEMP (Mattax and Quigley, 1989; Bartholow, 1991, 2000) and a model based on the HSPF (Hydrological Simulation Program – FORTRAN) model developed by the U.S. Environmental Protection Agency and the U.S. Geological Survey (Chen *et al.*, 1998a,b). Other models have been developed, but the ones mentioned are broadly representative of the range of complexity.

Sullivan *et al.* (1990) tested the ability of four reach scale models (Brown's model, TEMP-86, TEMPEST, and SSTEMP) and three catchment scale models (QUAL2E, SNTEMP, and MODEL-Y) to predict forestry related temperature increases in Washington. The catchment scale models required more input data than would be available for operational applications and did not provide accurate temperature predictions. TEMP-86 provided accurate predictions for mean, minimum, and maximum temperatures but required upstream temperatures as input to achieve the high level of performance. TEMPEST was less sensitive to specification of input temperatures, making it more suitable as an operational tool (Sullivan *et al.*, 1990).

Sridhar *et al.* (2004) addressed the problem of unknown upstream temperatures by using a reach length of 1,800 m above the prediction point. For this reach length, the effect of the upstream boundary condition on modeled downstream temperatures became negligible for low flow conditions. However, this approach would not necessarily be appropriate for the headmost streams in the channel network, where the reach of interest may extend only a few hundred meters or less downstream from the channel head. In such cases, an estimate of ground water temperature may be appropriate as an upstream boundary condition.

As mentioned previously, Rutherford *et al.* (1997) found that their model predictions were biased when the mean field measured values for shade were used as input. Although they were able to match the daily maximum and minimum temperatures by increasing the shade values to the maximum observed values, the timing of the diurnal temperature wave was incorrect, suggesting that some process was not properly represented. They hypothesized that flow through gravels (i.e., hyporheic exchange) could have been one of the causes. The significance of hyporheic exchange on reach scale temperature patterns should be investigated further.

DISCUSSION AND CONCLUSIONS

Summary of Forest Harvesting Effects on Microclimate and Stream Temperature

Forest harvesting can increase solar radiation in the riparian zone as well as wind speed and exposure to air advected from clearings, typically causing increases in summertime air, soil, and stream temperatures and decreases in relative humidity. Riparian buffers can help minimize these changes. Edge effects penetrating into a buffer generally decline rapidly within about one tree height into the forest under most circumstances. Solar radiation, soil temperature, and wind speed appear to adjust to forest conditions more rapidly than air temperature and relative humidity.

Clear-cut harvesting can produce significant daytime increases in stream temperature during summer, driven primarily by the increased solar radiation associated with decreased canopy cover but also influenced by channel morphology and stream hydrology. Winter temperature changes have not been as well documented but appear to be smaller in magnitude and sometimes opposite in direction in rain-dominated catchments. Although retention of riparian vegetation can help protect against temperature changes, substantial warming has been observed in streams with both unthinned and partial retention buffers. Road rights-of-way can also produce significant warming. Changes to bed temperature regimes have not been well studied but can be similar to changes in surface water in areas with downwelling flow.

Although the experimental results are qualitatively consistent, it is difficult to make quantitative comparisons of experimental results because the studies have expressed temperature changes using incommensurable temperature metrics. For the studies where similar metrics were available (e.g., maximum summer temperature), treatment effects exhibited substantial variability, even where the treatments appeared to be comparable (e.g., HJA Watershed 1 and Needle Branch). Thus, on their own, experimental results cannot easily be extrapolated to other situations. Application of heat budget models may help to diagnose the reasons for variations in response in experimental studies and provide a tool for confident extrapolation to new situations.

Increased stream temperatures associated with forest harvesting appear to decline to pre-logging levels within five to ten years in many cases, though thermal recovery can take longer in others. There is mixed evidence for the efficacy of low, shrubby vegetation in promoting recovery.

Temperature increases in headwater streams are unlikely to produce substantial changes in the temperatures of larger streams into which they flow, unless the total inflow of clear-cut heated tributaries constitutes a significant proportion of the total flow in the receiving stream. Clearing heated streams may or may not cool when they flow into shaded areas. Where downstream cooling does not occur rapidly, the spatial extent of thermal impacts is effectively extended to lower reaches, which may be fish bearing. In addition, warming of headwater streams could reduce the local cooling effect where they flow into larger streams, thus diminishing the value of those cool water areas as thermal refugia.

Biological Consequences and Implications for Forest Practices

It is difficult to estimate the biological consequences of harvesting related changes in riparian microclimate and stream temperature based on the existing results. In terms of terrestrial ecology in riparian zones, there is incomplete knowledge regarding the numbers of species that are unique to small streams and their riparian zones, as well as their population dynamics, sensitivity to microclimatic changes, and ability to recolonize disturbed habitat (Richardson *et al.*, 2005). The ecological effects of stream temperature changes in small, nonfish bearing streams are also unclear. While it is generally acknowledged that changes in thermal regime can influence macroinvertebrates (Vannote and Sweeney, 1980; Ward and Stanford, 1992), the metrics typically presented for stream temperature changes (e.g., maximum summer temperature) may not be the most biologically significant for streams that remain at sublethal temperatures. Given the emerging appreciation for the role of small streams in providing organic matter to downstream fish bearing reaches (e.g., Wipfli and Gregovich, 2002), a better understanding is required of how changes in the physical conditions in small streams and their interactions with chemical and biological processes influence their downstream exports.

Based on the available studies, a one-tree-height buffer on each side of a stream should be reasonably effective in reducing harvesting impacts on both riparian microclimate and stream temperature. Narrower buffers would provide at least partial protection, but their effectiveness may be compromised by wind throw, and they could still incur costs by complicating access and yarding operations. Alternative approaches to protecting riparian values may be possible that avoid at least some of the problems associated with buffers. For example, in B.C., many companies retain green tree patches within a cut block to provide future wildlife habitat. If these were positioned where they could shade the stream, they could provide at least some of the function of a riparian buffer but perhaps with lower wind throw risk and with less impact on ease of access and yarding.

Issues for Future Research

Riparian microclimates appear to have been relatively little studied, both in general and specifically in relation to the effects of different forest practices. Further research needs to address these knowledge gaps.

Shade is the dominant control on forestry related stream warming, and although algorithms exist for estimating it based on riparian vegetation height and channel geometry, there is a need to refine methods for measuring it in the field and for modeling it. Ground-based hemispherical photographs offer great potential for developing both static indices of shade as well as a tool for modeling the temporal variation of solar transmission as a function of the spatial distribution of canopy gaps. Further research should focus on the application of hemispherical photography, including an assessment of sampling variability and bias. In addition, the effects of low deciduous vegetation on the heat budget of small streams should be examined to help understand and predict trajectories of thermal recovery in time.

Further research should address the thermal implications of surface/subsurface hydrologic interactions. Studies should focus on both the local scale and reach scale effects of heat exchange associated with hyporheic flow paths, particularly those associated with step pool features, which are common in steep headwater streams. Bed temperature patterns in small streams and their relation to stream temperature should be researched, especially in relation to the effects on benthic invertebrates and other nonfish species. The hypothesis that warming of shallow ground water in clear-cuts can contribute to stream warming should be addressed, ideally by a combination of experimental and process/modeling studies.

The physical basis for temperature changes downstream of clearings needs to be clarified. In particular, it may be useful to determine whether diagnostic site factors exist that can predict reaches where cooling will occur. Such information could assist in the identification of "thermal recovery reaches" to limit the downstream propagation of stream warming. It could also help to identify areas within a cut block where shade from a retention patch would have the greatest influence.

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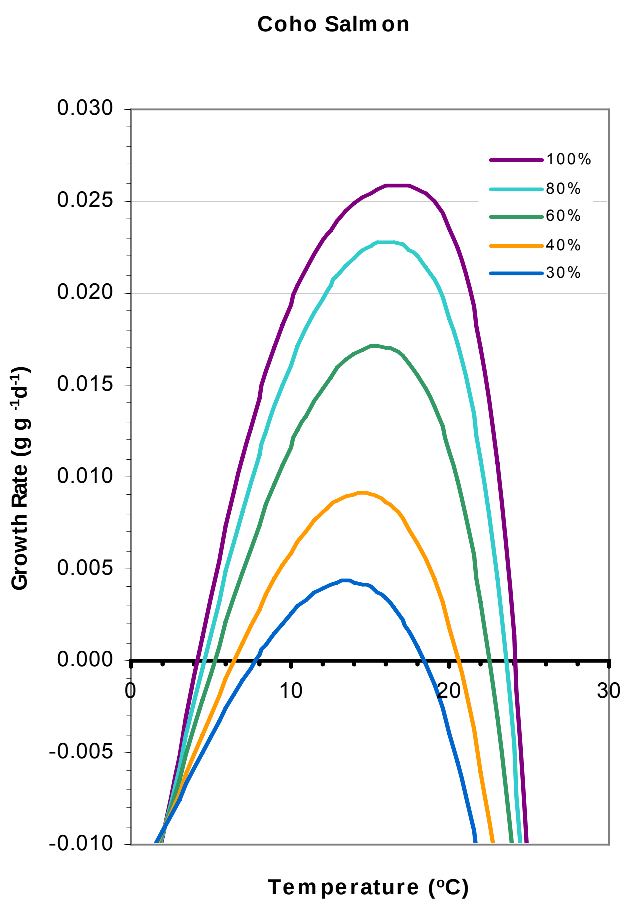
TAC Primer on The Physiological Basis For Salmonid Temperature Response and Watershed Pattern of Use

The Physiological Basis for Salmonid Temperature Response

Water temperature is a dominant factor affecting aquatic life within the stream environment (Hynes 1970). Water temperature affects important stream functions such as processing rates of organic matter, chemical reactions, metabolic rates of macro-invertebrates, and cues for life-cycle events (Sweeney and Vannote 1986). Water temperature plays a role in virtually every aspect of fish life, and adverse levels of temperature can affect behavior (e.g. feeding patterns or the timing of migration), growth, and vitality.

Figure 3. Coho salmon daily growth rate as a function of temperature and daily food ration.



Water temperature governs the rate of biochemical reactions in fish, influencing all activities by pacing metabolic rate (Frye 1971). Fish are poikilothermic or "cold-blooded". This means that fish do not respond to environmental temperature by feeling hot or cold. Rather, they respond to temperature by increasing or decreasing the rate of metabolism and activity. Water temperature is the thermostat that controls energy intake and expenditure.

The role of temperature in governing physiologic functions of salmonids has been studied extensively (Brett 1971; Elliott 1981; reviewed in Adams and Breck 1990; Brett 1995, McCullough 1999). The relationship between energetic processes and temperature have been quantified for many fish species with laboratory study.

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Energetic processes are expressed as functions of activity rate in relation to temperature. The relationships between energy-related functions and temperature follow two general patterns: either the rate increases continuously with rise in temperature (e.g., standard metabolic rate, active heart rate, gastric evacuation), or the response increases with temperature to maximum values at optimum temperatures and then decreases as temperature rises (e.g., growth rate, swimming speed, feeding rate) (Brett 1971, Elliott 1981). Each function operates at an optimal rate at some temperature and less efficiently at other temperatures.

For example, daily growth as a function of temperature is shown in Figure 1. Beginning with the coolest temperatures (0^{o} C), growth increases with temperature up to the optimal due to increasing consumption and food conversion efficiency. At temperatures above the optimal, growth rates decline as consumption declines in response to temperature and metabolic energy costs increase (Brett 1971, Elliott 1981, Weatherly and Gill 1995). Because the shape of growth curves is relatively broad at the maximum, there is little or no negative effect of temperature several degrees above optimum. Some investigators define the optimal temperature as the temperature at which maximum growth occurs, and refer to the range of temperature where growth occurs as “preferred” temperatures (Elliott 1981).

The general form of this relationship is similar for all salmonid species, varying somewhat in the details of growth rates and optimal temperatures. All salmonids have a similar biokinetic range of tolerance, performance, and activity. They are classified as temperate stenotherms (Hokanson 1977) and are grouped in the cold water guild (Magnuson et al. 1979). Significant differences in growth rate and temperature range exist among families of fish (Christie and Regier 1988). Some families grow best in colder temperatures (e.g. char), and many grow better in warmer temperatures (e.g. bass). Differences in the specific growth/temperature relationships among species in large measure explain competitive success of species in various temperature environments.

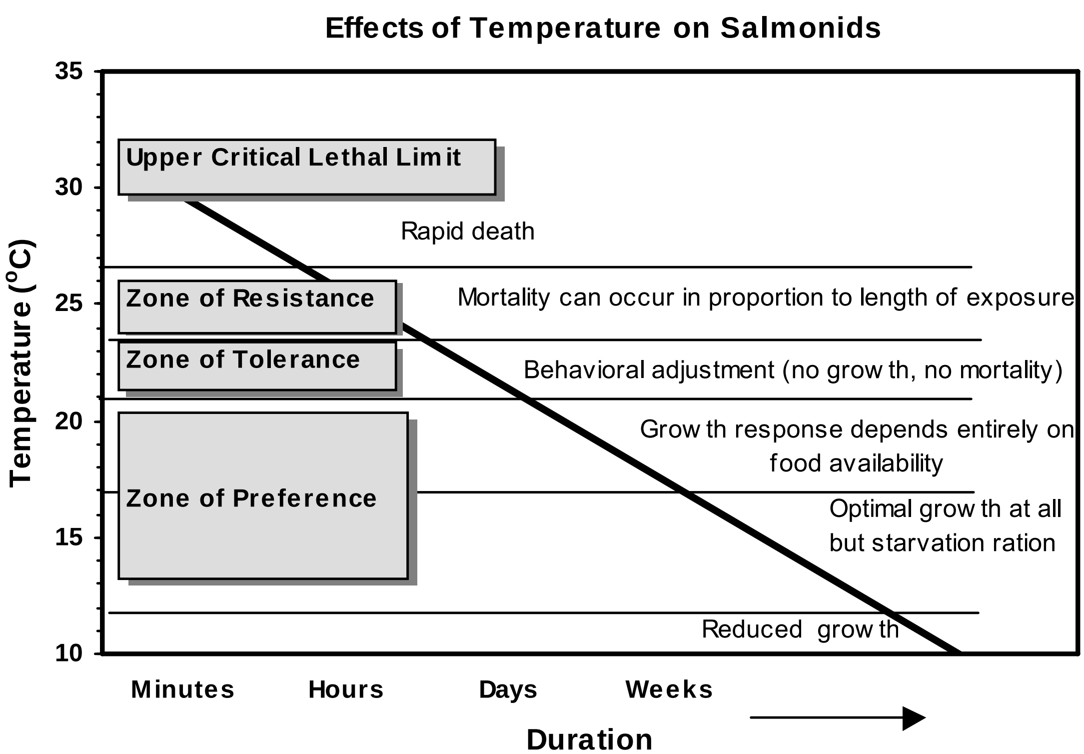
The range of environmental temperature where salmonid life is viable ranges from 0-30 ^{o}C, with critical temperatures varying somewhat by species. Salmonid physiologic functions operate most effectively in the mid regions of the range where growth is also optimized. Physiological functions are impaired on either end of the temperature range so that the geographic distribution of prevailing high or low temperatures ultimately limits the distribution of the species in the Salmonidae family (Eaton 1995).

The effects of temperature are a function of magnitude and duration of exposure. Figure 2 from Sullivan et al. 2000 summarizes the general relationship of salmonid response to temperature exposure. Salmon species are similar in this pattern, but vary somewhat in the temperatures zones of response.

Exposure to temperatures above 24°C can elicit mortality with sufficient length of exposure. The temperature where death occurs within minutes is termed the ultimate upper incipient lethal limit (UICL). This temperature is between 28- 30°C, varying by salmon species. Clearly, salmon populations are not likely to persist where this temperature occurs for even a few hours on a very few days each year (Eaton 1995). Lethal exposure is defined as up to 96 hours of continuous exposure to a given temperature.

Salmon can tolerate each successively lower temperature for exponentially increasing intervals of time. They do so by altering food consumption and limiting the metabolic rate and scope of activity (Brett 1971, Elliott 1981, Weatherly and Gill 1995). This resistance to the lethal effects of thermal stress enables fish to make excursions for limited times into temperatures that would eventually be lethal (Brett 1956; Elliott 1981). The period of tolerance prior to death is referred to as the "resistance time" (Figure 2) (Hokanson 1977, Jobling 1981). Salmon can extend their temperature tolerance through acclimation. Brett (1956) reported that the rate of increase in ability to tolerate higher temperatures among fish is relatively rapid, requiring less than 24 hours at temperatures above 20°C. Acclimation to low temperatures (less than 5°C) is considerably slower.

Figure 4. General biological effects of temperature on salmonids in relation to duration and magnitude of temperature (from Sullivan et al. 2000).



Laboratory and field studies have repeatedly found that salmon can spend very lengthy periods in temperatures between 22 and 24°C without suffering mortality (Brett 1995, Bisson et al. 1988; Martin 1988). Temperatures within this range may be stressful, but are not typically a direct cause of mortality (Brett 1956). Temperatures that cause thermal stress after longer exposures, ranging from weeks to months, are termed chronic temperature effects. Endpoints of lengthy exposure to temperature that are not physiologically optimum may include loss of appetite and failure to gain weight, competitive pressure and displacement by other species better adapted to prevailing temperatures (Reeves et al. 1987), change in behavior, or susceptibility to disease. Werner et al. (2001) documented correlations between stream temperature, size of juvenile steelhead and heat shock protein expression.

Fish may be able to avoid thermal stress by adjusting behavior, such as moving to cooler refugia. Numerous observers have observed behavioral adjustment by seeking cool water refugia when temperature in normal foraging locations reaches 22°C (Donaldson and Foster 1941; Griffiths and Alderdice 1972; Wurtsbaugh and Davis 1977; Lee and Rinne 1980; Bisson et. al. 1988; Nielsen et al. 1994, Tang and Boisclair 1995; Linton et al. 1997; Biro 1998). Fish resume feeding positions when temperatures decline below this threshold. At very low temperatures, salmonids cease feeding and seek cover under banks or within stream gravels (Everest and Chapman 1972).

Less quantifiable in a dose-response context are relationships involving temperature and disease resistance, and temperature effects on sensitivity to toxic chemicals and other stressors. (Cairns et al. 1978). For temperature to affect the occurrence of disease, disease-causing organisms must be present, and either those organisms must be affected by temperature or fish must be in a weakened state due to the effect of temperature. Some disease-causing organisms may be more prevalent at high temperature, others are more prevalent at low temperature, and some are not temperature-related. Thus, the interaction of temperature and disease is best evaluated on a location-specific basis.

If energy intake is adequate to fuel the physiological energy consumption, mediated in large part by the environmental temperature, then the organism can live in a healthy state and grow. Growth is a very important requirement for anadromous salmon living in fresh water. Salmon emerge from gravels in their natal streams measuring approximately 30 mm in length and weighing approximately 0.5 gram. Adults returning to spawn 3 to 5 years later typically measure 500 to 1000 mm in length and weigh from 5 to 20 kg depending on species. This enormous increase in body mass (greater than 5000 times) must be accomplished within a very limited lifespan. Salmon have evolved from a fresh water origin to spend a major portion of life in a marine habitat where there is far greater productivity and where the majority of growth occurs (Brett 1995).

Juvenile salmon must achieve the first six times increase in weight in their natal stream before they can smolt and migrate to the ocean (Weatherly and Gill 1995). Coho and steelhead generally smolt within 1 year, but can require as long as 3 years to achieve sufficient size to begin the transition to salt water. The long-term exposure of salmonids to temperature during their freshwater rearing phase has an important influence on the timing of smoltification and the ultimate size fish achieve (Warren 1971, Brett 1982, Weatherly and Gill 1995, Sullivan et al. 2000).

The size of salmonids during juvenile and adult life stages influences survival and reproductive success (Brett 1995). Larger size generally conveys competitive advantage for feeding (Puckett and Dill 1985, Nielsen 1994) for both resident and anadromous species. Smaller fish tend to be those lost as mortality from rearing populations (Mason 1976; Keith et al. 1998). Larger juveniles entering the winter period have greater over-wintering success (Holtby and Scrivener 1989; and Quinn and Peterson1996). Growth rates can also influence the timing when salmon juveniles reach readiness for smolting. Missing normal migration windows by being too small or too large, or meeting a temperature barrier, may have a negative effect on success in reaching the ocean (Holtby and Scriverner 1989).

How large a salmon can grow in a natural environment is fundamentally determined by environmental and population factors that determine the availability of food. Water temperature regulates how much growth can occur with the available food. Brett et al. (1971) described the freshwater rearing phase of juvenile salmon as one of restricted environmental conditions and generally retarded growth. Many studies have observed an increase in the growth and productivity of fish populations in streams when temperature (and correspondingly) food is increased. This tends to occur even in the cases where temperatures exceed preferred and sometimes lethal levels (Murphy et al. 1981, Hawkins et. al., 1983, Martin 1985, Wilzbach 1985, Filbert and Hawkins 1995).

Table 1 summarizes results from laboratory and field studies of coho and steelhead temperature response (from Sullivan et al 2000). Steelhead and coho are similar, though not identical, in the temperatures at which various functions or behaviors occur. Importantly, Sullivan et al (2000) showed that even though the laboratory optimal growth temperatures for steelhead are within a narrower and cooler range than those of coho (e.g. their "growth curves"), steelhead grow better than coho when exposed to higher temperatures in natural streams. These authors suggest that this disparity results from a greater efficiency in obtaining food in natural environments by steelhead, thus allowing them to generally obtain a higher ration of food. Bisson et al (1988b) showed that the body form of these two fish differ, enabling steelhead to feed efficiently in riffle habitats where food supply is more abundant. Thus, steelhead have a higher "net temperature tolerance" than coho.

With the exception of some spring-run Chinook salmon, most Chinook juveniles do not rear in streams through the summer and are therefore not typically exposed to late-summer conditions.

There has been some suggestion that there may be genetic adaptations by local populations that confer greater tolerance to temperatures. However, literature on temperature thresholds for salmonids, as summarized in Table 1 is remarkably consistent despite differences in locations of subject fish (Sullivan et al. 2000, Hines and Ambrose 2000, Welsh et al. 2001).

One problem encountered in synthesizing laboratory and field studies is how to characterize the widely variable stream temperature characteristics of a stream in either a physically or biologically meaningful way is lack of standardization on reporting summary statistics. The measures of 7-day maximum values have been shown to have biological meaning (e.g. Brungs and Jones 1977). These types of metrics also provide useful indices for comparing temperature among streams. Sullivan et al (2000) showed that all of the short-term high temperature criteria relate closely to one another when

Table 1. The spectrum of coho salmon and steelhead response at temperature thresholds synthesized for field and laboratory studies from Sullivan et al (2000). Threshold values are approximations, due to lack of consistency in reporting temperature averaging methods among studies. Temperature thresholds are standardized to the average 7-day maximum to the extent possible to allow comparison of field and laboratory study observations.

Biologic Response	COHO Approximate Temperature ^{o}C	STEELHEAD Approximate Temperature ^{o}C
Upper Critical Lethal Limit (death within minutes)-Lab	29.5	30.5
Geographic limit of species—Stream annual maximum temperature (Eaton 1995)	30	31.0
Geographic limit of species—Warmest 7-Day Average Daily Max Temperature (Eaton 1995)	23.4	24.0
Acute threshold U.S. EPA 1977—Annual Maximum	25	26
Acute threshold U.S. EPA 1977— 7-day average of daily maximum	18	19
Complete cessation of feeding (laboratory studies)	24	24
Growth loss of 20% (simulated at average food supply)	22.5	24.0
Increase incidence of disease (under specific situations)	22	22
Temporary movements to thermal refuges	22	22
Growth loss of 10% (simulated at average food supply) (7-day average of daily maximum)	16.5	20.5
Optimal growth at range of food satiation (laboratory)	12.5-18	10-16.5
Growth loss of 20% (simulated at average food supply) 7-day average of daily maximum	9	10
Cessation of feeding and movement to refuge	4	4

calculated from the same stream temperature record (7-day mean and maximum, annual maximum temperature, and long-term seasonal average). However, longer-term measures are better indicators of general ecologic metabolism. For example, degree-summation techniques sum duration of time (days, hours) above a selected threshold temperature.

Temperature Patterns and Salmonid Species Distribution Within Watersheds

Temperatures supporting the physiologic functions of fish species reflect the ambient temperatures likely to be found in streams in each species' natural range of occurrence (Hokanson 1977). For salmonids, this range is from 0 to less than 30^{o}C (see Table 1).

Within the range of distribution of salmonids in the Pacific Northwest, there is a west to east climatic gradient reflecting the marine influence at the coast and the orographic effects of interior mountain ranges. Coastal zones are characterized by maritime climates with high rainfall that occurs during the winter and dry warm summers. Interior zones are dryer, and rainfall may occur as rain or snow. Summers are very dry, and temperatures often hotter than coastal zones, although elevation can have a significant cooling effect. Comparison of river temperatures associated with forested regions throughout Washington, Oregon and Idaho show generally consistent occurrence of temperatures within the temperature tolerance of salmonids (Sullivan et al. 2000).

The temperature of streams and rivers within the range of distribution of salmonids in the Pacific Northwest and California typically vary widely on both temporal and spatial scales. For example, the range of hourly temperature over a year period for a smaller headwaters stream and larger mainstem river located within a forested watershed in Washington are shown in Figure 3. (The figure also shows the typical phase and migration timing for coho and steelhead salmon.) Similar patterns are observed in forested regions of California.

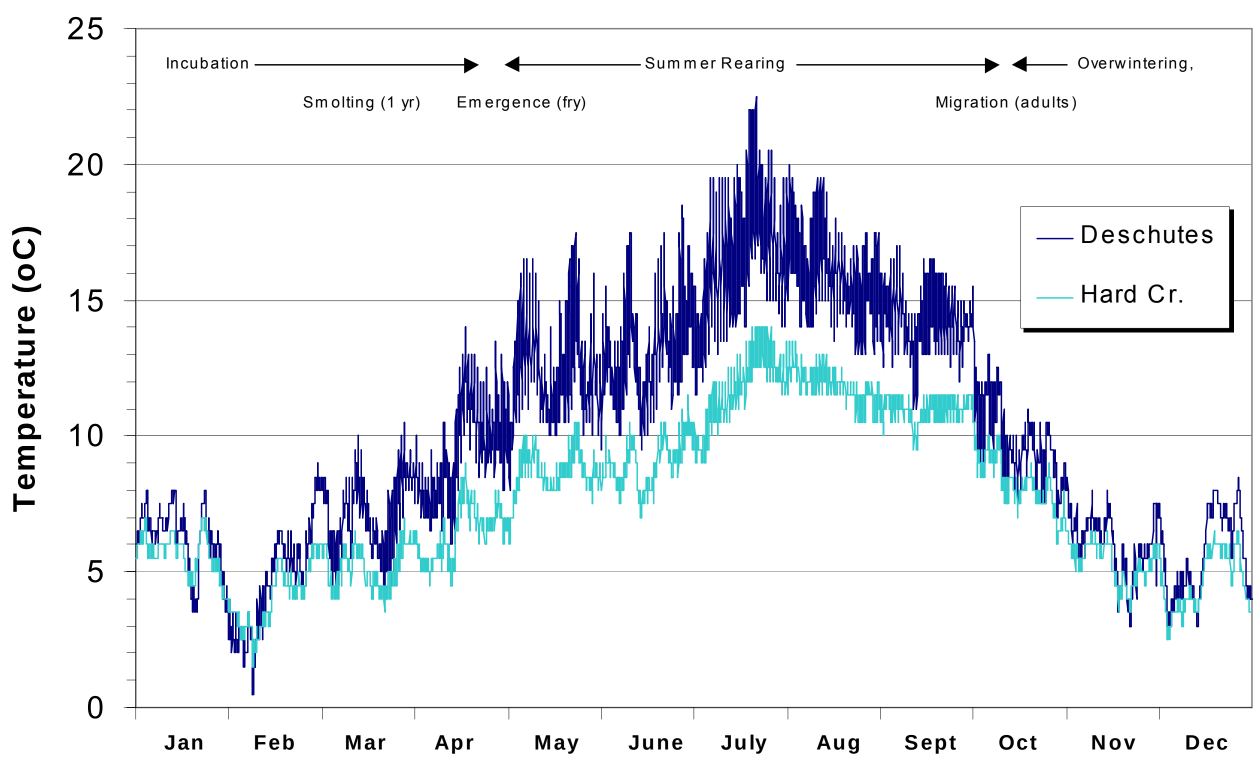
Active feeding and positive growth can occur at any time during the year when temperature is within the positive growth range illustrated in Figure 1. Juvenile salmon experience preferred temperatures for much of the year, and may experience stressful temperature conditions for relatively little time during the year. Water temperatures between 8 and 22^{o}C tend to be the most prevalent temperatures observed in natal rivers and streams in the Pacific Northwest (Sullivan et al. 2000). Temperatures high enough to directly cause mortality are rare within the region where salmon occur. Temperatures high enough to cause stress (>22^{o}C) may be common, especially in higher order streams.

Watershed Temperature Patterns

Stream temperature tends to increase in the downstream direction from headwaters to lowlands. (Hynes 1970, Theurer et al 1984). The dominant environmental variables that regulate heat energy exchange for a given solar loading, and determine water temperature are stream depth, proportional view-to-the-sky, rate and temperature of groundwater inflow, and air temperature (Moore et al, 2005). Increasing temperature in the downstream direction reflects systematic tendencies in these critical environmental factors. Groundwater input becomes a smaller portion of the streamflow and has less cooling effect as streams get larger (Sullivan et al 1990). Air temperature increases with decreasing elevation (Lewis et al. 2000). Riparian vegetation and topography shade a progressively smaller proportion of the water surface as streams widen (Spence et al. 1996), until at some location there is no effective shade at all (Beschta et al. 1987, Gregory et al. 1991). Streams gain greater thermal inertia as stream flow volume increases (Beschta et al. 1987), thus adjusting more slowly to daily fluctuations in energy input. The typical watershed temperature pattern is illustrated in Figure 4.

Water temperature in larger rivers without riparian shading is in equilibrium with, and close to, air temperature. In smaller streams, water temperature is depressed below air temperature due to the cooling effects of groundwater inflow and the shading effects of the forest canopy (Sullivan et al. 1990; Poole and Berman 2000, Moore 2005). The minimum temperature profile in Figure 4 indicates the general pattern of water temperature in streams in a fully forested watershed. The coolest temperatures will be

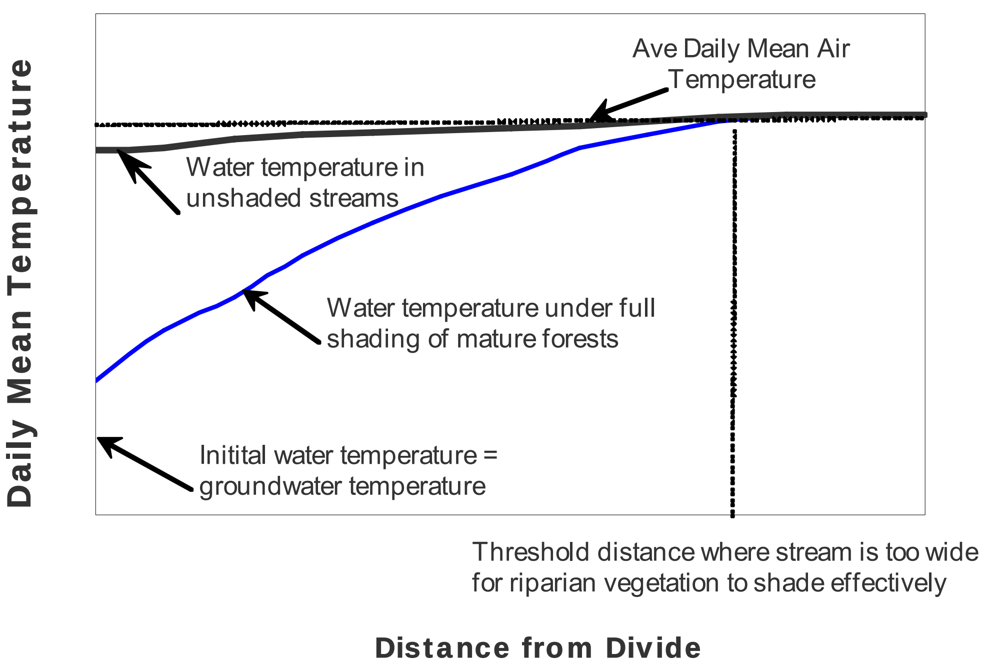
Figure 5. Water temperature of the Deschutes River (148 km^2) and Hard Creek (2.3 km^2), a headwater tributary in the Cascades of Washington. Data are hourly measurements.



observed in the smallest streams and will be near prevailing groundwater temperature. As the effects of these insulating variables lessens in the downstream direction, water temperature moves closer to air temperature until the threshold distance where riparian canopy no longer provides effective shade and the water temperature is closely correlated with air temperature alone (Kothandaraman 1972). It is likely that the shape of the minimum line varies both with basin air temperature and with differences in natural vegetation.

Various authors have reported the likely summertime temperatures that mark the highest and lowest temperatures on this curve for streams and rivers of the Pacific Northwest and California used by salmonids. Minimum groundwater temperatures are approximately 10-13^{o}C (Sullivan et al. 1990, Lewis et al. 2000). Maximum temperatures typically range from 20 to 26^{o}C (Sullivan et al. 2000, Lewis et al. 2000) depending on location.

Figure 4. General pattern of temperature at the watershed scale and potential range of response to forest removal. (from Sullivan et al. 1990).



Removal of vegetation in headwater streams may allow temperature to increase up to (but not exceed) the basin air temperature maxima. Thus, the potential response of water temperature to forest harvest may be large in small streams, but only small, and difficult to detect in mid to large size watersheds.

Fish Species Distribution Within Watersheds

Salmonid species found in California include Chinook (O. tshawytscha), coho (O. kisutch), and steelhead (O. salmo). These species are the most temperature tolerant of the anadromous species in the salmonidae family. The southern-most extent of the natural range of salmon is found at latitude approximately equal to San Francisco, dipping further south along the coast. Eaton (1995) showed a strong relationship between prevailing summertime maximum temperatures and the end of the range of occurrence.

Salmon species throughout their range have evolved to use different parts of the river system during their freshwater rearing phase.   Systematic changes in the occurrence or dominance of species within river systems in part reflects the temperature patterns as one important component of habitat. Differences among species can confer competitive advantages in relation to environmental variables that influence the species' distribution (Brett 1971, Baltz et. al. 1982, Reeves et al. 1987, DeStaso and Rahel 1994).

Steelhead have higher net temperature tolerance, are widely distributed within the northern region of California and occupy a broader range of habitats including larger rivers and smaller streams. Coho have the lowest net temperature tolerance of the salmonids found in California, and are found primarily where temperatures are coolest for most of the year. They primarily occur in the low to mid-order tributaries within the coastal zone.

Chinook salmon are perhaps the most temperature tolerant of all salmon species. They have the highest optimal temperatures for growth and fastest growth rates of all the salmonids. Fall run chinook emerge from gravels in spring and move to the larger (warmer) rivers where their growth rate allows them to migrate to the ocean with weeks to a few months. The juveniles migrate out of the river before the warmest summer temperatures occur.

An exception are spring-run Chinook salmon. Some juveniles reside in streams throughout the summer. These salmon are also the only salmonid that must cope with summer water temperatures as adults. They typically enter the Sacramento River from March to July and continue upstream to tributary streams where they over-summer before spawning in the fall (Myers et al. 1998). Adult spring-run Chinook salmon require deep, cold pools to hold over in during the summer months prior to their fall spawning period. When these pools exceed 21^{o}C adult Chinook salmon can experience decreased reproductive success, retarded growth rate, decreased fecundity, increased metabolic rate, migratory barriers, and other behavioral or physiological stresses (McCullough 1999).

California Regional Temperatures

To date, there has been no California-wide water temperature study or synthesis of available information. A regional stream temperature study was conducted within the Coho ESU by the Forest Science Project at Humboldt State University (Lewis et al. 2000). The area where coho occur within California is delineated by the Coho ESU includes the northern coast zone and portions of the interior Klamath region. The regional study measured water temperature at hundreds of sites in a variety of streams and rivers well distributed within the area from approximately San Francisco northward to the Oregon border, and from the coast to approximately 300 km inland. Stream size varied from watershed areas as small as 20 to a maximum of over 2,000,000 hectares. The assessment included new data and historical analysis of historic temperature assessments, augmented with recently measured temperature at the same locations as earlier measurements.

Results of the study provide some general insight into maximum summer stream temperatures within this region of California.

1) The regional study confirmed the general increasing trends in temperature from watershed divide to lowlands.

2) The annual maximum temperature ranged from 12-25^{o}C in the coastal zone and 14-32^{o}C inland beyond the coastal influence. Temperature as high as 32^{o}C occurs, but is rare.

3) The cooling influence of the coastal fog belt on air temperature extends as far inland as 50 km in some rivers, and is significant enough to affect water temperature within a distance 20 km from the coast in some locations. The effect of the cool air is sufficient to reduce some river temperatures by as much as 5-7^{o}C degrees by the time water reaches the ocean. These help prevent prolonged exposure to stressful temperatures. The coast fog zone is the dominant zone for coho productivity in the state.

4) Maximum temperature in rivers in the coastal fog belt can still exceed 20^{o}C

5) No one geographic, riparian, or climatic factor explains water temperature with high precision. Multiple regression models developed from the data explain about 65% of the variability, similar to finding in other parts of the Pacific

Northwest (Sullivan et al. 1990).

6) The coolest maximum temperatures (<18^{o}C) are most likely to occur where:

- Distance from divide is less than 10 km.
- Canopy cover is >75%

7 The probability of achieving temperature of <20^{o}C decreases at 1) lower canopy closure, 2) distance from divide as an indicator of stream size, and 3) with distance from the coast.

8 There is relatively small difference in maximum water temperatures between interior and coastal streams of similar watershed areas in basins less than 100,000 hectares in size.

What needs to be understood better for California:

1) the availability of cool water at the watershed and population scale

2) the overall cumulative effect of temperature on the annual basis.

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KS6/15/07

KEY QUESTIONS: HEAT AND MICROCLIMATE

Embedded or implied in each key question below are the following issues for the contractor's synthesis:

a. Relationship to each of California's regions;
b. Context for riparian buffer strip size: stream order, stream class, topography, hydrologic regime, and climate;
c. Context for comparisons: pristine, 'optimum', legacy, or pre-harvest conditions
d. Relationship of the quality of forest management practices being evaluated to current California forest practices; ability of BMP's to effectively mitigate identified problems;
e. Relationship of temperature alterations to salmonid habitat quality

How do forest management activities or disturbances within the riparian area affect the temperature of forest streams?

a. What conditions of canopy structure, density, and width, influence water temperature? How might this vary with California forest types and stream size?

b. Are riparian area microclimates affected by forest management within and/or adjacent to fish-bearing streams sufficient to influence water temperature?

c. How and to what extent do temperatures in low order streams influence temperatures in downstream fish-bearing streams?

How and where are the potential temperature effects from forest management likely to impact salmonid species of concern?

a. Is there information from California eco-regions indicating the effects of observed temperature on salmonids?

b. Are there conditions that adequately ameliorate the occurrence of adverse temperatures?

What bearing do the findings of this literature review have on riparian zone delineation or characteristics of riparian zones for protecting water temperature?

INITIAL LIST OF LITERATURE for Contrtactor Review:

Heat and Microclimate

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Appendix D: Sediment Riparian Exchange Function
Primer, Key Questions and Initial List of Literature to be reviewed.

Primer
on
Sediment
Riparian Exchanges Related to Forest Management in the Western U.S.

Prepared by the
Technical Advisory Committee
of the
California Board of Forestry and Fire Protection

May 2007

Version 1.0

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PRIMER: SEDIMENT RIPARIAN EXCHANGE FUNCTION

Erosion and Sediment Processes in California's Forested Watersheds

Erosion is a natural process that is well described for California in several college textbooks (Norris and Webb 1990, Mount 1995). California's evolving landscape reflects the "competing processes of mountain building and mountain destruction", with landslides, floods, and earthquakes working as episodic forces which often create major changes (Mount 1995). In general, the land surface is sculpted by the forces of erosion: water, wind, and ice. The physical and chemical composition of the rock determines how it weathers by these forces. The role of running water in shaping the earth's surface is considered the most important of all the geologic processes and has received the greatest attention by researchers (Leopold et al. 1964; Morisawa 1968).

The rates of natural erosion are very high in the State's regions having greater amounts of rain and snow, such as the geologically young mountains of the Northern Coast Ranges, Klamath Mountains, and Sierra Nevada (Norris and Webb 1990). Mean annual precipitation was shown to be a relatively precise indicator of climatic stress on sedimentation in Northern California (Anderson et al. 1976).

Soil erosion processes on upland watersheds include: a) surface erosion (e.g., dry ravel, sheet and rill), b) gullying, and c) mass movement or wasting (e.g., soil creep and landslides, such as slumps, earthflows, debris slides, large rotational slides). These can occur singly or in combination. Falling raindrops can be a primary cause of surface erosion, especially where soils have little vegetative cover (Brooks et al. 1991). Erosion products deposited by water become "sediment", brought to a channel by gravity and erosive forces. The water-related, or "fluvial", processes active within the stream channel and floodplain are: 1) the transport of sediment; 2) the erosion of stream channel and land surface; and 3) the deposition or storage of sediment.

Sediment Sizes, Transport & Measurement

Sediment is any material deposited by water, but research usually describes sediment according to its size, means of transport, and method of measurement (MacDonald et al. 1991, Leopold 1994). Inorganic sediment ranges in size from very fine clay to very large boulders. Particle size classes tend to be split into a different number of size categories by physical scientists (AGI 2006) and by biologists (Cummins 1962). The Modified Wentworth Scale is commonly used by biologists (Waters 1995) and includes 11 particle sizes and names: clay, silt, sand (five classes), gravel, pebbles, cobbles, and boulders. In addition, sediment includes particulate organic matter, composed of organic silts and clays and decomposed material. Grain size terminology can also vary:

- *Fine-grained sediment* ("fines") includes the smaller particles, such as silt and clay (usually <0.83 mm in diameter). The largest size class for this category varies, sometimes including sand and small gravel (1-9 mm) (Everest et al. 1987).
- *Coarse-grained sediment* represents the larger particles, such as gravels and cobbles. It makes up the bed and bars of many, if not most, rivers. The smallest size class for this category varies, and sometimes includes sand and small gravel (1-9 mm).

Whatever the term used, it is important to understand the sediment definition and particle size that each research article is using before extrapolating the results.

Sediment is transported by streams as either *suspended load* of the finest particle sizes (from clay to fine sand <2.0 mm) that are carried within the water column, or as *bedload* of the larger particles (from coarse sand to boulders) that never rise off the bed more than a few grain diameters. Higher velocity and steeper streambed slope can transport larger grain size, for example.

Since the measurement of sediment transport levels can be problematic, it is done in several ways. (For detailed descriptions of common methods, including the strengths and limitations of each, see MacDonald et al. 1991, Gordon et al. 1992, and Waters 1995.)

Suspended sediment samplers measure direct suspended sediment concentration (SSC) in milligrams of sediment per liter of water (mg/l). Since most sediment transport takes place during high flows, samples must be taken during these periods to develop long-term averages. Many samples are needed near peak discharges to determine the error margin. Two types of samplers can be used: depth-integrating and point-integrating.

Turbidity is a measure of the ability of light to be transmitted through the water column (e.g., the relative cloudiness). Turbidity sampling and meters are often used as a substitute for the direct measurement of the suspended sediment load of a selected stream reach, but the relationship may vary and requires a careful study design to make accurate correlations Turbidity is frequently higher during early season runoff and on the rising limb of a storm's runoff; automated data collection is now being used to more accurately capture such infrequent events (Eads and Lewis 2003). Turbid water may also be due to organic acids, particulates, plankton, and microorganisms (which can be ecologically beneficial); interpretation must therefore be carefully done. In redwood-dominated watersheds of north coastal California, Madej (2005) found the organic content of suspended sediment samples ranged from 10 to 80 weight percent for individual flood events. Turbidity is not a good indicator for movement of coarse-grained sediments, such as sand in granitic watersheds, since these larger grain sizes move at the bottom of the water column or as bedload (Morisawa 1968; Sommarstrom et al. 1990; Gordon et al. 1992).

Bedload measurement can be a difficult method since this larger-sized sediment must be collected manually during high flows when bedload is in transport. While there are different types of methods and equipment, the Helley-Smith bedload sampler has become the standard for bedload measurement, especially for coarse sand and gravel beds. Multiple samples must be taken per cross-section of stream. Bedload cannot be collected automatically as readily as suspended sediment can. Bedload as a percentage of suspended load can range from 2-150 percent; 10 percent bedload would be a conservative estimate for a storm event with muddy-looking water in a gravel-bed stream.

Sediment that is deposited within stream channels can be measured by changes in channel characteristics. The most common methods include: a) channel cross-sections, b) channel width / width-depth ratios; b) pool parameters (e.g., fines stored in pools (V*)), c) bed material (particle-size distribution, embeddedness, surface vs. subsurface particle size); d) longitudinal profiles in upstream-downstream directions (e.g., using the "thalweg", the deepest part of the stream channel).

Fluvial Processes and Sediment

Stream reaches can be defined by the dominant fluvial processes: erosion /transport / storage (Schumm 1977; Montgomery and Buffington, 1997; Bisson, *et al,* 2006). The steep headwaters tend to be the source of erosion, the middle elevation streams are the transfer zone, and the low elevation streams are the depositional zone. However, any given stream reach demonstrates all three processes over a period of time; the relative importance varies by location in the watershed.

Natural Sources of Sediment

Within the riparian zone, natural sediment sources and the effects of the riparian zone tend to vary by the type of channel reach (Montgomery and Buffington, 1997; Bisson, *et al,* 2006). The uppermost parts of many source reaches are characterized by exposed bedrock, glacial deposits, or colluvial valleys or swales. Stream reaches in bedrock valleys are usually strongly confined and the dominant sediment sources are fluvial erosion, hillslope processes, and mass wasting. The colluvial headwater basins have floors filled with colluvium which has accumulated over very long periods of time. Such channels as may exist are directly coupled with the hillslopes, and their beds and banks are composed of poorly graded colluvium. Stream flow is shallow and ephemeral or intermittent. The colluvial fill is periodically excavated by debris flows which scour out the stream channels and delivery large quantities of sediment and large woody debris to downstream reaches (Montgomery and Buffington, 1997; Bisson, *et al,* 2006). There is often is no distinctively riparian vegetation bordering the channels.

A bit further downstream, transport reaches commonly still have steep gradients, are strongly confined and subject to scouring by debris flows. Stream beds are consequently characterized either by frequent irregularly arranged boulders or by channel-spanning accumulations of boulders and large cobbles that separate pools. The boulders move only in the largest flood flows and may have been emplaced by other processes (e.g., glacial till, landslides). Streams generally have a sediment transport capacity far in excess of the sediment supply (except following mass wasting events). Dominant sediment sources are fluvial and hillslope processes and mass wasting (Montgomery and Buffington, 1997; Bisson, *et al,* 2006). The transition between transport and response reaches is especially likely to have persistent and pronounced impacts from increased sediment supply (Montgomery and Buffington, 1997).

In the higher response reaches, stream gradients and channel confinement become more moderate. Incipient floodplains or floodprone areas may begin to border the channels, so they are not so coupled to hillslope processes. The typical channel bed is mostly straight and featureless with gravel and cobble distributed quite evenly across the channel width; there are few pools. Where the bed surface is armored by cobble, sediment transport capacity exceeds sediment supply, but unarmored beds indicate a balance between transport capacity and supply. Dominant sediment sources are fluvial processes, including bank erosion, and debris flows are more likely to cause deposition than scouring (Montgomery and Buffington, 1997; Bisson, *et al,* 2006). There is usually distinctively riparian vegetation along the channel.

Also in low to moderate gradients, braided reaches may form where the sediment supply is far in excess of transport capacity (e.g., glacial outwash, mass wasting) and/or stream banks are weak or erodible (Buffington, *et al*, 2003). Channels are multi-threaded with numerous bars. The bars and channels can shift frequently and dramatically, and channel widening is common. The size of bed particles varies widely. Banks are typically composed of alluvium. Bank erosion, other fluvial processes, debris flows, and glaciers are the dominant sediment sources. Distinctively riparian vegetation is common, and is especially important in providing root strength to weak alluvial deposits (Bisson, *et al*, 2006).

In lower-elevation, lower-gradient response reaches, channels are generally sinuous, unconfined by valley walls, and bordered by floodplains. Beds are composed of gravel or sand arranged into ripples or dunes with intervening pools. Sediment supply exceeds sediment transport capacity, so much of the finer sediment is deposited outside the channel onto the floodplain. The dominant sediment sources are fluvial processes, bank erosion, inactive channels, and debris flows. Distinctively riparian vegetation typically grows on the floodplain where it plays important roles in: i) reinforcing weak alluvial banks and floodplains, and ii) providing hydraulic roughness to reduce erosion during overbank flooding (Montgomery and Buffington, 1997; Bisson, *et al,* 2006).

Natural sediment production in undisturbed watersheds can vary significantly, depending upon soil erodibility, geology, climate, landform, and vegetation. Delivery of sediment to channels by surface erosion is generally low in undisturbed forested watersheds, but can vary greatly by year (Swanston 1991). Annual differences are caused by weather patterns, availability of materials, and changes in exposed surface area. Sediment yields for surface erosion tend to be naturally higher in rain-dominated than in snow-dominated areas. Soil mass movement is the predominant erosional process in steep, high rainfall forest lands of the Pacific Coast. The role of natural disturbances in maintaining and restoring the aquatic ecosystem is becoming more recognized by scientists using interdisciplinary approaches (Reeves et al. 1995).

California Examples

Landslides are an important sediment source in northern coastal ranges of California, particularly where they were active in the wet period of the late Pleistocene and have remained dormant for long periods. If reactivated by undercutting at the toe, these slides can deliver immense amounts of sediment to channels (Leopold 1994). Kelsey (1980) found in the Van Duzen River basin that avalanche debris slides accounted for headwater erosion storage, but that natural fluvial hillslope erosion rates were quite low. In the North Coast range, small headwater streams tend to aggrade their beds during small storms and degrade during large, peak flow events. However, in larger streams, sediment aggrades during large events and gradually erodes during smaller ones (Janda et al.1978).

Sediment budgets offer a quantitative accounting of the rates of sediment production, transport, storage, and discharge (Swanson et al. 1982; Reid & Dunne 1996). They are performed in California by academic researchers (Kelsey 1980; Raines 1991), consultants (e.g., Benda 2003), and agencies. In a review of sediment source analyses completed for agency-prepared Total Maximum Daily Load (TMDL) allocations in nine north coast California watersheds, the amount of the “natural” sediment source contribution ranged from a low of 12% to a high of 72% over the past 20-50 year period (Kramer et al. 2001). An evaluation of sediment sources in a granitic watershed of the Klamath Mountains found 24% of the erosion and 40% of the sediment yield to be natural background levels in 1989 (Sommarstrom et al. 1990). Post-fire erosion can be a major component of sediment budgets in semi-arid regions of California (Benda 2003).

Role of Riparian Vegetation

Forested riparian ecosystems influence sediment regimes in many ways. First, riparian plant species are adapted to flooding, erosion, sediment deposition, seasonally

saturated soil environments, physical abrasion, and stem breakage (Dwire et al. 2006). Sediment transported downslope from overland flow passes by riparian vegetation, where it can accumulate or be transported through the riparian area (USEPA 1975; Swanson et al. 1982b). The significance of vegetation's role in providing bank stability and improving fish habitat was first recognized as early as 1885 (Van Cleef 1885). Riparian plant roots help provide streambank, floodplain, and slope stability (Thorne 1990; Abernathy and Rutherford 2000; NRC 2002) and can bind bank sediment, reducing sediment inputs to streams (Dunaway et al. 1994). Bank material is much more susceptible to erosion below the rooting zone, but vegetated banks are typically more stable than unvegetated ones (Hickin 1984). Soil, hydrology, and vegetation are interconnected in bank stability, though the understanding has developed more slowly (Sedell and Beschta 1991; NRC 2002). For example, the effect of riparian vegetation roots on the mass stability of stream banks may be overestimated in erosion models, according to recent research (Pollen and Simon 2005). In a study on the Upper Truckee River, California, a willow species provided an order of magnitude more root reinforcement than lodgepole pine and reduced the frequency of bank failures and sediment delivery (Simon, Pollen, and Langendoen 2006).
Riparian vegetation patterns appear to indicate specific landforms and local hydrogeomorphic conditions; the patterns differ by geographic location and climate, such as semi-arid versus humid regions (Hupp and Ostercamp 1996). Since streamside areas tend to have high moisture and low soil strength, they are vulnerable to compaction and physical disturbance (Dwire et al. 2006). For some sediment processes originating from upslope of the riparian zone, vegetation may have little influence. Large, deep-seated landslides are probably not affected by streamside plants and downed wood, for example (Swanson et al. 1982b). Current conditions of riparian plant communities need to be viewed in the context of the historical alterations to the landscape, including land management (NCASI 2005).

Effects of Sediment on Aquatic Life of Streams

While erosion processes can provide sources of gravels for fish spawning, excessive sediment deposition can be harmful to aquatic life. Habitat needs for anadromous salmonid fish of the Pacific Coast are well described by Bjornn and Reiser (1991), with a review of the effects of fine sediment on fish habitats and fish production compiled by Everest et al. (1987), Furniss (1991), Walters (1995), Spence et al. (1996), and CDFG (2004). A brief summary of the effects of sediment on critical life stages of salmon and trout is as follows:

- Spawning: Fine sediment can become embedded in spawning gravels, reducing the abundance and quality available for spawning and possibly preventing the female from excavating her nest (redd); excessive sediment loading can cause channel aggradation, braiding, widening, and increased subsurface flows, all reducing spawning gravel abundance; excess sediment can fill pools that are needed for rest and escapement of adults migrating upstream to spawn.

- Egg Incubation: Excessive fine sediments can suffocate or impede egg development or developing alevins by reducing or blocking intragravel water flow, oxygenation, and gas exchange. Organic sediment, however, can provide valuable food (e.g., bugs) for fish (Madej 2005).
- Juvenile Rearing: Coarse and fine sediment can fill pools, which reduces the volume of habitat available for critical rearing space and the population that can be sustained; fine sediment can cover the streambed and suffocate benthic macroinvertebrates, reducing availability of important food source (Suttle et al. 2004). Chronic turbidity from suspended fine sediment interferes with feeding effectiveness of fry and smolts, reducing their growth rate or forcing them to emigrate (Sigler et al. 1984; Newcombe and Jensen 1996; Rosetta 2004).

The review by Everest et al. (1987) demonstrated that the effects of fine sediment on salmonids are complex and depend on many interacting factors: species and race of fish, duration of freshwater rearing, spawning escapement within a stream system, presence of other fish species, availability of spawning and rearing habitats, stream gradient, channel morphology, sequence of flow events, basin lithology, and history of land use (Furniss et al. 1991). It also should be noted that research on the effect of “fine sediment” on salmonid reproduction (e.g., percent survival of fry emergence from eggs) varies in the definition of sediment size, ranging from 0.85mm to 9.5 mm, but tends to focus on 2.0 millimeters or less (Everest et al. 1987). One needs to be careful in interpretation of the literature when comparing the effects of differently defined “fines” (Sommarstrom et al. 1990.)

The first major literature review on the aquatic effects of human-caused sediment was published in 1961 by California Dept. of Fish and Game biologists Cordone and Kelley, who concluded that sediment was harmful to trout and salmon streams. Productive streams, at every trophic level, contain stored sediment and large organic debris and are more productive than channels with too little or too much sediment (Everest et al. 1987). An early California study of streams with increased sedimentation found that fish biomass decreased in some streams and increased in others (Burns 1972). Stream macroinvertebrate diversity was significantly decreased in stream reaches below failed logging road crossings, implying the effect of higher sediment levels (Erman et al. 1977). In a review of stream characteristics in old-growth forests, the authors noted that many streams in California have naturally high sediment loads, including an abundance of fines less than 1 mm, but historically these streams supported healthy populations of salmonids (Sedell and Swanson 1984).

Forest Management & Sediment Effects

The literature on the erosion and sediment impacts of forest operations is quite extensive, though much of it comes out of the Pacific Northwest. Most of the California research on private forestland has focused on the north coastal redwood region, particularly in the Caspar Creek Experimental Watershed of the Jackson Demonstration State Forest in Mendocino County (e.g., Zeimer 1998; Rice et al. 2004) and in the Redwood Creek watershed as part of Redwood National Park related research (e.g., Best et al. 1995; Madej 2005).

Historic Logging Practices

Certain mid-20th century logging practices were clearly identified as harming water quality. Clearcut logging, of large portions of a watershed down to the edge of streams, and the logging road system, were noted as a major source of sediment in earlier studies in Oregon (Brown and Krygier 1971; Swanson and Dyrness 1975) and California (Cordone and Kelly 1961; Burns 1972). Cordone and Kelley in 1961 perceived that the bulk of stream damage was caused by carelessness and could be prevented "with little additional expense", they thought at the time. Over thirty years ago, Burns (1972) examined logging and road effects on juvenile anadromous salmonids in northern California streams, with all streams showing sediment increases following logging. Evidence was also gathered to show that good logging practices could reduce sedimentation problems in the western region (Haupt and Kidd 1965; Brown 1983).

Sediment and other impacts led to a series of increasingly protective measures for forestry operations on public and private lands in the U.S. In 1973, California's State Water Resources Control Board recommended improved timber harvest and road construction methods at the time of the passage of the State Forest Practice Act but prior to the adoption of the Forest Practice Rules in 1975 by the Board of Forestry (SWRCB 1973). Tighter stream protection rules were later required by the State, as described under Riparian Buffers below. Berbach (2001) describes the evolution of such measures for private forestland in California.

Roads as a Major Source of Sediment

Logging roads have historically been the largest, or one of the largest, sources of forest management-related sediment (Trimple and Sartz 1957; Megahan and Kidd 1972; Burns 1972; Anderson et al. 1976; Adams & Ringer 1994). One study found that roads can contribute more sediment per unit area than that from all other forestry activities, including log skidding and yarding (Gibbons and Salo 1973). Roads can affect streams directly through the acceleration of erosion and sediment loadings, the alteration of channel morphology, and changes in the runoff characteristics of watersheds. Sedimentation was often greatest when major storm events occurred immediately after construction, while surface erosion usually declined over time with revegetation of roadsides and natural stabilization (Beschta 1978). A long-term study in Caspar Creek in Mendocino County found similar results, but also a lag of sediment transport as material only moved during periods of high runoff and streamflow (Krammes and Burns 1973). In landslide prone terrain, road-related erosion could continue unless certain design, construction and maintenance practices were carried out, or high erosion hazard areas were avoided. Much of the research of logging road effects was on roads that had been constructed in the 1950's, 60's and 70's, before improved road location and design to minimize potential slope stability and erosion problems were applied. By the early 1990s, steps were being taken to minimize the negative effects of roads on streams through both construction and maintenance practices (Furniss et al. 1991; Weaver and Hagans 1994).

Channel crossings, within the riparian area, are often the primary cause of water quality problems associated with roads and the resultant ecological impacts (USFS 1976; Erman et al. 1977; Forman and Alexander 1998). Debris blockages of undersized culverts and flood flows can cause the failure of the logging road stream crossing, delivering large volumes of crossing-fill sediment directly into the channel. In a long-term erosion evaluation of the Redwood Creek watershed, researchers found significant gullying problems due to logging roads, particularly due to diversions at plugged stream culverts or ditch relief culverts (Hagans et al. 1986). These diversions created complex channel networks and increased downslope drainage density, yet 80% of all gully erosion was avoidable, the authors stated, through minor changes in road construction techniques.

Heavily used, unsurfaced logging roads also can produce significantly more sediment and turbidity than abandoned roads, with one study in Washington State showing a 130 fold increase (Reid and Dunne 1984). Road surface sediment can drain into roadside ditches and then into streams, delivering fine sediment detectable by turbidity sampling below the road (Bilby et al. 1989). The problem can be effectively minimized, the authors noted, by draining the ditch onto the forest floor in small quantities to infiltrate, by using better road construction and surfacing material, and by leaving woody debris within the stream. Ketcheson and Megahan (1996) evaluated the potential sediment filtration effectiveness of the riparian zone below road fills and culverts in granitic terrain, finding that road sediment travel distance increased with increasing volume of eroded material.

In some locations, road placement within the stream riparian zone can encroach on the floodplain and channel and force streamflows to the opposite bank, potentially destabilizing the hillslope and causing increased landsliding. Roads located within the landslide-prone inner valley gorge, where very steep slopes are adjacent to streams, are at high risk of frequent or iterative failure (Furniss et al. 1991). A study in the Klamath Mountains of northwestern California noted this relationship (Wolfe 1982). If roads must be located in a valley bottom, a buffer strip of natural vegetation between the road and the stream is recommended (Furniss et al. 1991).

High quality roads and better maintenance are likely to reduce the amount of material supplied to channels from hillslopes, reduce the amount of sediment mobilized along low order streams, and reduce the sediment delivery rate to high order streams (Furniss et al. 1991; Slaymaker 2000). In the past decade, methods to inventory logging road drainages for their potential to deliver sediment have become more standardized (Flanagan et al. 1998; CDFG 2006). Road erosion studies need to be examined in the context of geology and soil types, such as the highly erosive granitics (e.g., Megahan and Kidd 1972).

Some studies have compared the effects of old to new forest practices. Cafferata and Spittler (1998) compared the effects of logging in the 1970s to the 1990s in the Caspar Creek watershed in Mendocino County found that “legacy” roads continue to be significant sources of sediment deca des after construction. Recent Total Maximum Daily Load (TMDL) studies in north coastal California watersheds assessed sediment sources over multiple decades, but the analyses did not distinguish whether logging road-related sediment originated from roads constructed before or after the Forest Practice Act in 1973 (Kramer et al. 2001). However, timber operations under the “modern” Forest Practice Rules produced an estimated erosion rate one-tenth that of pre-1976 practices on a tributary of Redwood Creek (Best et al. 1995). Rice (1999) cautioned about direct comparisons of different studies with different objectives, but concluded that road-related erosion in Redwood Creek was significantly reduced due to improved road standards (e.g., better sizing and placement of culverts). In 1999, the Scientific Review Panel on California Forest Practice Rules and Salmonid Habitat made nine recommendations on road construction and maintenance, including the removal of legacy roads within the riparian zone (Ligon et al. 1999).

Riparian Buffers in Forest Management

The concept of using vegetation and/or obstructions to form buffer strips to minimize or retard downslope sediment movement has been applied to agricultural and forestry operations for many years (Broderson 1973; USEPA 1975). Buffer strips are defined as riparian lands maintained immediately adjacent to streams or lakes to protect water quality, fish habitat, and other resources (Belt et al. 1992). Limiting mechanical harvesting activities within streamside zones is appropriate to protect their vulnerability to compaction and physical disturbance, due to high moisture and low soil strength factors (Dwire et al. 2006).

The U.S. Forest Service adopted the Streamside Management Zone (SMZ) in the 1970s as a Best Management Practice (BMP), for closely managed harvesting, to act as an effective filter and absorptive zone for sediment, to protect channel and streambanks, and other benefits (USFS 1979). Each National Forest's Forest Plan also has Standards and Guidelines for the protection of riparian areas, including specific BMPs (Belt et al. 1992). In 1975, the California Board of Forestry first adopted the Stream and Lake Protection Zone (SLPZs) as part of the state's Forest Practice Rules (FPRs); these riparian zone protections were later expanded by the Watercourse and Lake Protection Zone (WLPZ) in 1983, 1991 and 2000 (Berbach 2001). While the benefits of such riparian protections are not challenged, the extent of the buffer strips (i.e., upslope and upstream) to balance ecological, water quality, and management needs continues to be debated (Dwire et al. 2006).

Direct physical disturbance of stream channels and soils within the riparian area by timber harvest activities can increase sediment discharge (Everest et al. 1987). In a 1975 California field study, physical damage to streambanks during logging was caused by equipment operating through streams, by yarding and skidding timber through channels, and by removal of streamside vegetation. Failed road crossings deposited sediment into the streams, reducing the diversity of the aquatic invertebrate community (Erman et al. 1977). Grant (1988) identified a method, primarily through aerial photograph analysis, to detect possible downstream changes in riparian areas due to upstream forest management activities.

More recent studies have looked at the design of forest riparian buffer strips to protect water quality. The authors of one literature summary stated, "we cannot overemphasize the importance of maintaining the integrity of the riparian zone during harvest operations" in relation to erosion and sedimentation processes (Chamberlin et al. 1991). The use of riparian buffers and BMPs has generally decreased the negative effects of forest harvest activities on surface water quality (Belt et al. 1992; Norris 1993). However, even an intact riparian buffer strip cannot prevent significant amounts of hillslope sediment from entering a stream via overland flow (due to infiltration and saturation excess in severely disturbed soil) or from debris slides originating outside the riparian zone (Belt and O'Laughlin 1994; O'Laughlin & Belt 1995).

One area of research receiving more attention is the riparian zone within headwater and low order streams (e.g., first and second). Sediment deposited in low order streams (which tend to be Class III under FPR rules) may be delivered to high order streams (e.g., third and fourth) that are usually Class I and II. Moore (2005) summarizes the latest results of this headwater research in the Pacific Northwest. MacDonald and Coe (2007) have recently investigated the influence of headwater streams on downstream reaches in forested areas, including the connectivity and effects of sediment. These recent research papers and others on this topic need to be thoroughly examined before consensus can be reached on the conclusions.

In recent years, the use of riparian buffer zones as a management tool has increased. For public lands in the Pacific Northwest, Riparian Reserves (RR) were set aside under the Northwest Forest Plan in 1994, where silvicultural activities were not allowed for multiple reasons, including water quality (Thomas 2004). For private forest lands, stream protection zones have increased in importance and restrictions in the past decade due to the federal and state listings of anadromous salmonid species as threatened or endangered (Blinn and Kilgore 2001; Lee et al. 2004). The current WLPZ rules for California were tightened from the 1991 Rules to protect listed fish species under the "Threatened or Impaired" (T/I) Rules, adopted as Interim Rule Requirements by the BOF in 2000, based in part on the recommendations of the Scientific Review Panel (Ligon et al. 1999; Berbach 2001). Research is now needed on the effects of these newer riparian protection zones, with comparisons made to previously designated zones.

Recent Sediment Evaluations of Forest Practices

Evaluations of forest practices producing and delivering sediment, as a nonpoint pollution source, revealed that Best Management Practice (BMP) implementation was generally good across the U.S., but cases of noncompliance persisted (especially for road and skid trail BMPs (SWRCB 1987; Binkley and Brown 1993). The authors recommended compliance and effectiveness monitoring must therefore be an ongoing activity.

The Board of Forestry's Monitoring Study Group (MSG) has overseen two recent evaluations of the effectiveness of the Board's Forest Practice Rules (FPRs). The Hillslope Monitoring Program (Cafferata and Munn 2002) evaluated monitoring results from 1996 through 2001, while the Modified Completion Report (Brandow et al. 2006) continued analysis of data from 2001 through 2004. Both studies found that: 1) the rate of compliance with the FPRs designed to protect water quality and aquatic habitat is generally high, and 2) the FPRs are highly effective in preventing erosion, sedimentation and sediment transport to channels when properly implemented. The 2006 report concluded the following:

> In most cases, Watercourse and Lake Protection Zone (WLPZ) canopy and groundcover exceeded Forest Practice Rule (FPR) standards. With rare exceptions, WLPZ groundcover exceeds 70%, patches of bare soil in WLPZs exceeding the FPR standards are rare, and erosion features within WLPZs related to current operations are uncommon. Moreover, in most cases, actual WLPZ widths were found to meet or exceed FPR standards and/or widths prescribed in the applicable THP…
>
> When properly implemented, road-related FPRs were found to be highly effective in preventing erosion, sedimentation and sediment transport to channels. Overall implementation of road-related rules was found to meet or exceed required standards 82% of the time, was marginally acceptable 14% of the time, and departed from the FPRs 4% of the time. Road-related rules most frequently cited for poor implementation were waterbreak spacing and the size, number and location of drainage structures…
>
> Watercourse crossings present a higher risk of discharge into streams than roads, because while some roads are close to streams, all watercourse crossings straddle watercourses. Overall, 64% of watercourse crossings had acceptable implementation of all applicable FPRs,

> while 19% had at least one feature with marginally acceptable implementation and 17% had at least one departure from the FPRs. Common deficiencies included diversion potential, fill slope erosion, culvert plugging, and scour at the outlet…

Attention has recently focused on riparian management of low order streams by management agencies, the public, and scientists. Gaps in knowledge are still being identified for the Pacific region and the diversity of riparian management standards continue to be debated (Young 2000; Moore 2005).

What We Do Not Know or Do Not Yet Agree Upon:

- The need for buffer strips along low order (e.g., 1st, 2nd) streams to prevent or minimize the delivery of sediment to higher order streams during forestry operations.
- The amount of forest management that can be performed within a designated riparian buffer zone without accelerating sediment production and delivery.
- The sediment effects of the newer, riparian protection zones for forest management, with comparisons made to previously designated zones.
- The relevance of forest management research on sediment relationships in riparian zones in other western states to California, and the relevance of such research in California's north coastal redwood region to other region's of the state.

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SS4/17/07

KEY QUESTIONS: SEDIMENT

Much research has occurred on the relationship of forest management practices to sediment production and delivery (see Sediment Primer). Although roads and watercourse crossings have been identified as a primary sediment source, their impacts are not the focus of this BOF-TAC effort except where appropriate within the scope of the following key questions. Seeking to resolve the remaining uncertainties related to forest management effects on sediment and the riparian zone is the emphasis of this investigation.

Embedded or implied in each key question below are the following issues for the contractor's synthesis:

A. Relationship to each of California's regions;
B. Context for riparian buffer strip size: stream order, stream class, topography, hydrologic regime, and climate;
C. Context for comparisons: pristine, 'optimum', legacy, or pre-harvest conditions;

D. Relationship of the quality of forest management practices being evaluated to current California forest practices; ability of BMPs to effectively mitigate identified problems;
E. Relationship of sediment alterations to salmonid habitat quality and feeding effectiveness.

1) How do forest management activities or disturbances in or near the riparian zone affect the <u>production</u> of sediment over space and time?

a) To what extent and with what mechanisms are zero and low-order streams (e.g., first- and second-order) and their riparian zones a significant source of sediment production in unmanaged and managed forest areas?
b) How effective are current forest management practices in or near the riparian zone in mitigating the production of sediment in higher-order streams (e.g., third-order and higher)?
c) To what extent and in what ways is sediment production from channels, streambanks and flood-prone areas affected by current forest management practices? Does plant succession stage or vegetative community have any effect?

2) How do forest management activities or disturbances in or near the riparian zone affect the <u>delivery</u> and storage of sediment over space and time?

a) To what extent and with what mechanisms are zero and low-order streams (e.g., first- and second-order) a significant source of sediment delivery in unmanaged and managed forest areas?
b) How effective are current forest management practices in mitigating the delivery of sediment in higher-order streams (e.g., third-order and higher)?
c) To what extent and in what ways is sediment delivery from channels and streambanks and storage on flood-prone areas affected by current forest management practices? Does plant succession stage or vegetative community have any effect?
d) Are there forest practices that can remobilize the sediment deposited within the riparian zone and flood-prone areas and redeliver into the stream system?
e) How effective are riparian buffer zones in providing a sediment filtering function in unmanaged and managed forest areas?

3) Based on the results of the above, what riparian zone <u>delineation</u> or <u>characteristics</u> (e.g., cover, plant species and structure, etc.) are shown to be needed to ameliorate sediment production and delivery from managed forests?

a) Is there a threshold or degree of effectiveness based on benefit (e.g., channel and streambank stability, upslope filtration, surface stability in floodprone areas, sediment storage due to hydraulic roughness)?

b) How does effectiveness vary by geographical region, geology, size of watershed, vegetation, stream reach, forest practices within and nearby the zone, etc.?

c) What are the types of erosion events for which buffer zones are not effective in preventing or reducing sediment delivery and those for which they are relatively effective?

SS 3/23/07

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Appendix E: Water Riparian Exchange Function

Primer, Key Questions and Initial List of Literature to be reviewed.

Primer on Water Riparian Exchanges Related to Forest Management in the Western U.S.

Prepared by the
Technical Advisory Committee
of the
California Board of Forestry and Fire Protection

May 2007

Version 1.0

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Salmonid Life-Cycle Needs Related to Water

Important habitat characteristics for salmonids in streams include minimum streamflow, obstructions to flow that create debris dams and have other effects on stream shape, and gravel necessary for spawning (Botkin and others 1994). The riparian zone along streams influences all of these factors. Streamflow, and the sediment this flow transports, interact with large wood, boulders, and bedrock outcrops to produce physical characteristics of streams required by fish, including side channels in floodplains, and pools and riffles in small main-stream channels.

The amount, velocity, and depth of water required by salmonids varies depending on the life stage. Bjornn and Reiser (1991) present a comprehensive review of this topic for North American salmonids. Migrating fish require water depths that allow upstream passage [e.g., minimum water depths of 0.09 m to 0.12 m for chum salmon, depending on substrate particle size (Sautner and others 1984)]. Streamflow affects the amount of spawning habitat available by regulating the area covered by water and the velocities and depths of water over gravel beds [e.g., velocities ranging from 0.3 to 3.0 m/s and a minimum depth of 0.18 m (Thompson 1972)]. Stream discharge, followed by water velocity, are the most important factors in determining the amount of suitable living space for rearing salmonids [e.g., velocities < 10 cm/s for newly emerged salmon and trout fry (Everest and Chapman 1972); depths ranging from water barely deep enough to cover juveniles to > 1 m (Bjornn and Reiser 1991)].[1] In general, salmonid carrying capacity increases as streamflow increases up to a point, and then levels off or declines if velocity becomes excessive (Bjornn and Reiser 1991, Murphy 1995).

Minimum streamflows in both summer and late fall are critical for juvenile rearing and successful spawning for salmonids, respectively. Murphy (1995) reported that minimum streamflow in summer limits salmonid carrying capacity on a broad scale. For example, total commercial catch of coho salmon off of Washington and Oregon was found to be directly related to the amount of summer streamflow when the juveniles were in streams two years before (Smoker 1955, Mathews and Olson 1980). Botkin and others (1994) found that streamflow, especially the minimum flow in November three and four years prior to adult returns, accounted for most of the variation in adult spring Chinook adult salmon returning to spawn in the Rogue River in Oregon.

Effects of Forest Management on Peak Flows, Low Flows, and Water Yield

The effects of forest management activities on streamflow have been studied since the early 1900's and are summarized in Ziemer and Lisle (1998) and Moore and Wondzell (2005). Changes in peak flows, low flows, and water yield

[1] Note that in an area with numerous deep pools and cool groundwater contribution, discharge and velocity can be very low, compared to an area without pools.

resulting from forest removal are very complex. The magnitude of change to both water yield and peak flows depends on the amount and location of the harvest, the stand age and composition of the vegetation removed, soil and lithologic characteristics, topography, and climatic conditions. The persistence of the effect is largely determined by the rate and composition of vegetation re-occupying the disturbed site.

In terms of aquatic habitat, key hydrologic concerns relate to changes in summer low flows, and in peak flows and their effects on channel stability and sediment transport (Moore and Wondzell 2005). In a comprehensive review of forestry impacts on aquatic habitats, Botkin and others (1994) concluded that there is no evidence or reason to believe that changes in flow due to forest harvest would be deleterious to fish. They state that increases in flood peaks would be expected to cause a slight increase in channel mobility and an increase in the transport of bed sediment (factors that relate to spawning and rearing habitat), but there do not appear to be field studies relating changes in flooding to degradation of fish habitat.

Peak Flow Changes

Ziemer and Lisle (1998) provide a comprehensive description of how changes in peak flows associated with forest management vary with watershed size, type of precipitation, season, and flood magnitude. In general, the effects of forest practices are more pronounced and easier to detect in small watersheds, greater in areas where rain-on-snow events occur, greater in the fall months, and greater for frequent runoff events. More detailed information on these principles and specific examples and are provided in the paragraphs that follow.

Substantial (e.g., $\geq$30-50% clearcut) harvesting in small to medium-sized watersheds[2] over short time periods is required to noticeably increase small to medium recurrence-interval peak flows associated with timber harvesting. Limited harvesting in riparian areas alone cannot affect flood frequency or magnitude.

Ziemer (1998) reported a 9 percent increase in 2-year peak flows following clearcutting approximately 50 percent of the North Fork Caspar Creek watershed (5 km^2), located near Fort Bragg, California.[3] Ziemer and Lisle (1998) state that: "There is little evidence that forest practices significantly affect large floods produced by rain. However, it is possible that clearcutting exacerbates some rain-on-snow floods, although the magnitude of such an effect is highly variable

[2] Ziemer and Lisle (1998) define small basins as having drainage areas $\leq$ 1 km^2 (~250 ac) and large basins as >100 km^2 (~25,000 ac). Medium-sized basins can be considered be on the order of 10 km^2 (~2,500 ac).
[3] The WLPZ Forest Practice Rules tested in the North Fork Caspar Creek watershed were those in effect from 1983 to 1991 (e.g., Class I buffer strips of 200 ft for slopes >70%). In 1991, maximum Class I WLPZs were reduced to 150 feet for slopes >50%.

and difficult to measure or detect."[4] They also explain that the greater the size of the flood or basin being investigated, the less likely that there will be any detectable changes caused by forest practices.

Specific peak flow studies in the Pacific Northwest confirm these conclusions. Thomas and Megahan (1998) found that treatment effects decreased as flow event size increased and were not detectable for flows with 2-year return intervals or greater for small treated watersheds that were either clearcut or patchcut with roads in the H.J. Andrews Experimental Forest, located in the western Cascade Mountains of Oregon in the rain-on-snow zone. Beschta and others (2000) analyzed the same data and concluded that treatment effects were unlikely for peak flows with recurrence intervals of approximately 5 years or greater, and that a relationship could not be found between forest harvesting and peak discharge in the large basins.

In a broad summary of the literature, Moore and Wondzell (2005) reported that peak flows increased following forest harvesting in most studies in coastal catchments, with increases ranging from 13 percent to over 40 percent based on the original analyses. They also found that in coastal watersheds, the magnitude of forest practice-related peak-flow increases declined with increasing event magnitude in most cases, with the greatest increases typically associated with autumn rain events on relatively dry catchments. Moore and Wondzell (2005) state that peak flow change does not appear to be related in any simple way to the percentage of basin area cut or basal area removed, and that estimates of post-treatment recovery rates varied among studies.

Timber harvesting affects the amount of interception loss that takes place in forested watersheds. This, in turn, may influence changes in winter peak flows. Interception loss has been reported as approximately 20% in coastal California forests (Reid and Lewis, in press), and more generally as about 10 to 30 percent of total rainfall, depending on canopy characteristics and climatic conditions (Moore and Wondzell 2005). Differences in interception loss between logged and unlogged areas are likely to explain the majority of the observed increases in larger winter peak flows, when transpiration is at its annual minimum (Ziemer 1998, Lewis and others 2001).

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[4] Snow accumulation tends to be higher in openings than under forest canopies, with cut blocks typically accumulating about 30 percent to 50 percent more snow. Removal of the forest canopy exposes the snow surface to greater incident solar radiation as well as to higher wind speeds, which can increase sensible and latent heat inputs. During mid-winter rain-on-snow events, melt rates are typically governed by sensible heat transfer from the relatively warm air, condensation of water vapor onto the snowpack, and in some cases by the sensible heat of rainfall. Under these conditions, snowmelt may significantly augment rainfall, increasing the magnitude of flood peaks (Moore and Wondzell 2005).

Small increases in peak flows (≤ 10%) for 2-5 yr return interval events have been found to be relatively benign and have not been judged to be capable of substantially modifying the morphology of the stream channels (Ziemer 1998). This is due to the fact that the magnitude of peak flow changes is substantially less than the within-a-year and year-to-year variability in streamflows. The changes are within the normal range of variability of streamflows (Grant and others 1999).

In addition to harvesting effects, roads can have significant hydrologic impacts (Coe 2004). Several studies have shown that logging roads can intercept shallow subsurface flow and rapidly route it to the stream network, potentially leading to increased peak flows in headwater basins (Moore and Wondzell 2005), or possibly delayed peaks in larger watersheds due to desynchronization of peak flows from tributary basins. Pathways linking the road network to stream channels include roadside ditches draining directly to streams, and roadside ditches draining to culverts that feed water into incised gullies (Wemple and others 1996). Accelerated runoff at the road segment scale also results since haul roads have compacted surfaces with low permeability that generate overland flow in even moderate rainstorms (Coe 2004, Moore and Wondzell 2005).

At the basin scale, paired-watershed studies have not shown strong evidence to support road-induced increases in peak flows. Studies may have been hampered by insufficient pre-treatment calibration data, lack of treatment replication, and poor experimental control (i.e., road building and timber harvesting have often occurred simultaneously or in quick succession) (Thomas and Megahan 1998, Coe 2004). Modeling studies have shown that increases in peak flows due to roads were approximately equal to the effects from timber harvesting (i.e., canopy removal) in an experimental watershed in western Washington (Bowling and Lettenmaier 2001). The effect of both activities declined as the flow recurrence interval increased. Additionally, modeling studies suggest that roads can decrease baseflow during the critical summer months (Tague and Band 2001). However, much uncertainly still exists regarding the hydrologic effects of roads at the watershed scale (Coe 2004, Royer 2006). If there are impacts from road building on peak flows, these effects will be more pronounced and easier to detect in smaller basins (Ziemer and Lisle 1997).

Channel aggradation, or filling of the channel bed with sediment, can have a significant effect on flood height or flooding. Where aggradation is severe, it is more important for overbank flooding than changes in runoff due to logging operations (Lisle and others 2000). Widespread channel aggradation can occur in low gradient reaches of watersheds if the sediment production rate has been significantly accelerated above background rates by mass wasting and surface erosion and delivery processes. If this happens, similar magnitude peak flows to those which would have occurred earlier can cause more extensive over-bank flooding downstream because of reduced channel capacity. These flood events would be the consequence of rainfall/runoff/channel aggradation interactions, rather than rainfall/runoff interactions. The area flooded would be changed by the altered channel configuration, even if the amount of water remained the same.

Low Flow Changes

Forest removal in mountainous watersheds will increase low summer and early fall streamflows, as well as total water yield. Botkin and others (1994) reported that while total water flow in a stream is important to salmon, flow increases during summer and early fall that can augment streamflow at a critical season for juvenile rearing are more important than the changes in magnitude of total annual flow. Nearly all published reports on timber harvesting and resulting changes in summer low flows have shown that streamflow will either increase or remain unchanged in proportion to the amount of vegetation removed in the watershed. Harvested areas contain wetter soils than unlogged areas during periods of evapotranspiration, and hence higher groundwater levels and greater late-summer streamflow (Chamberlin and others 1991).

Studies have documented that the post-treatment recovery rates are highly variable depending on the severity of the treatment and the vegetation reoccupying the site, along with physiographic and climatic characteristics. Often increases are fairly short-lived, as regeneration begins to utilize surplus soil moisture and intercepts precipitation. After approximately 10-30 years, baseflow (and peak flow rates) have returned to normal or decreased below pre-harvest levels due to rapidly growing hardwoods that transpire more water than mature conifer trees (Murphy 1995, Moore and Wondzell, 2005). Long-term effects of logging on summer low flows likely depends primarily on species composition before and after harvest (Spence and others 1996, Moore and Wondzell 2005). In general, summer low flows are more sensitive to transpiration from riparian vegetation than from vegetation in the rest of the catchment (Moore and Wondzell 2005).

One example in California of documented water yield changes with both selective harvesting and clearcutting has taken place in the Caspar Creek watershed. The effects of selective logging on low flows were examined in the South Fork Caspar Creek watershed, where 64 percent of the second-growth stand volume of coast redwood and Douglas-fir was tractor logged from 1971 to 1973. Statistically significant summer low flow enhancements were evident for 7 years after logging. Minimum discharge increases averaged 38 percent after the selective harvesting and summer low flow volumes increases averaged 29% between 1972 and 1978 (Keppeler and Ziemer 1990, Rice and others 2004). The average length of the part of the low flow period when flow in the South Fork was less than 0.2 cfs was shortened by 43 days form 1972 to 1978, a 40% reduction. As in previous studies, most of the enhanced streamflow (average annual water yield) increase (approximately 90 percent) was realized during the rainy season while greater relative increases were witnessed during the summer low flow period (Keppeler 1986).

In the North Fork Caspar Creek watershed, approximately 50 percent of the watershed was clearcut harvested over about 7 years (1985 to January 1992).[5] Minimum discharge increases averaged 148 percent at the North Fork weir and flow enhancement persisted through hydrologic year 1997 with no recovery trend observed. The larger increases in the North Fork were probably due to wetter soils in the clearcut units, where little vegetation was present to use the additional moisture (Keppeler 1998). This data suggests that water yield effects will persist longer after clearcutting than when a similar timber volume is removed from a watershed with selective cutting. These differences in water yield recovery are probably related to changes in rainfall interception and evapotranspiration (Rice and others 2004). Enhanced summer low flows improve aquatic habitat in stream channels. In the Caspar Creek study, higher discharge levels increased habitat volumes and lengthened the flowing channel network along logged reaches during the summer and early fall months (Keppeler 1998).

The amount of increased water flow caused by forest management activities on summer low flows of large rivers is unknown, but Botkin and others (1994) state that based on studies extrapolated elsewhere, it is reasonable to assume that there would be a small positive effect. Given the importance of low flow increases to salmonid production, however, this change may be significant.

Annual Water Yield Changes

For total annual water-yield changes with forest management, most small-watershed studies have shown that in areas with significant precipitation (>100 cm/yr or ~40 in/yr), increases in streamflow are proportional to the reduction in forest cover. This is due to reduced losses from evapotranspiration by the trees in rain-dominated systems. Moore and Wondzell (2005) reported that in rain-dominated small catchments, clearcutting and patch-cutting increased yields by up to 6 mm for each percentage of basin harvested, while selective cutting increased yields by up to about 3 mm for each percentage of basal area removed. Increased water yield, however, is not uniformly distributed seasonally or throughout the rotation in the Pacific Northwest and California. Most of the annual increase occurs in the winter high-runoff season and during the wetter years, rather than during the summer season and drought years, when the additional water is needed (Ziemer 1987).[6] When vegetation reduction in a

[5] Most of the clearcut harvesting (45.5%) took place from the spring of 1989 to January 1992 (Henry 1998).

[6] This was observed in areas with rain-dominated winter periods, where summer storms are infrequent, as is found in California. In contrast, experimental studies on eastern U.S. watersheds (rain-dominated) have shown that peakflow and water yield increases dominate

watershed is less than 20 percent, the expected water-yield increase is not measurable and the remaining trees will likely use as much water as the original stand (Bosch and Hewlett 1982).

Ziemer (1987) summarized the literature on this subject and reported that total water yield increases resulting from management in larger basins would be very small and not measurable. For example, Kattelmann and others (1983) estimated that for National Forest lands in Sierra Nevada watersheds, streamflow could only be increased one percent if multiple use/sustained yield guidelines were followed.

While there is some evidence in the arid southwestern United States that expansion of the phreatophytic riparian forests along rivers can contribute to streamflow declines (Thomas and Pool 2006), this does not appear to be a significant concern for most California watersheds with coniferous forests. For forest streams with narrow strips of riparian forest, riparian vegetation water use is usually a small portion of the overall water budget and probably has minor influence on annual water yield (Dr. Julie Stromberg, Arizona State University, Tempe, AZ, personal communication). As an example, complete felling of a strip of riparian vegetation in a small watershed at Coweeta Hydrologic Laboratory in North Carolina produced only very minor water yield increases (Hewlett and Hibbert 1961). With the limited harvesting in riparian zones that is allowed under the current forest practice rules in California, water-yield increases are not expected to be measurable.

Stormflow Generation

Water is transferred through riparian zones to channels by surface and subsurface flow. Shallow or lateral subsurface flow from hillslopes in steep forested watersheds in the western United States is widely recognized as a main contributor to stream flow generation; however, processes that control how and when hillslopes connect to streams are still being studied. Much of the difficulty in deciphering hillslope response in the stream is due to riparian zone modulation of these inputs (McGuire and McDonnell 2006).

A key concept for forested watersheds is that there is great temporal and spatial variability in how water is transferred to the channel. Streamflow in small forested headwater basins is usually generated from an expanding and contracting source area, often denoted as the variable source area, representing a fraction of the total basin area. The source of streamflow is usually that part of the basin nearest the perennial, intermittent, and ephemeral channels. Source areas (the hydrologically-active areas that contribute directly to stormflow) can vary from only one percent of the total basin area in small storms to 50 percent or more in very large storms. The percentage of saturated source area in a

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during the growing season months, since approximately half of the annual precipitation (in the form of higher-intensity convective storms) occurs from May through October.

watershed is topographically sensitive (i.e., higher percentages occur with gentler slopes). The source areas within a watershed are very dynamic, expanding and contracting during events as the influx of precipitation progresses and then ends.

Moisture redistribution continues following the rain event as slower lateral hillslope drainage supplies additional moisture to lower slope positions. Direct runoff and its source area increase due to channel expansion and slope water movement (Hewlett and Nutter 1970, Troendle 1985). Riparian areas associated with perennial and larger intermittent streams remain at or near saturation during the winter and hence are hydrologically active for transporting water by saturated overland flow and rapid subsurface flow via soil macropore and/or displacement flowpaths. Smaller intermittent and ephemeral streams are only active when the hydrologic network expands sufficiently to incorporate steeper-gradient channels. Ephemeral first order channels (typically Class III watercourses) flow only in response to direct rainfall, and, although they are part of the hydrologic network, they do not generally have riparian zones because hydrophilic (water-dependent or water-loving) plants are usually absent.

Water Exchange and Transfer within the Riparian/Floodplain Zone

Water is exchanged in riparian zones, and larger floodplains in several ways. Streams either gain water from inflow of groundwater (i.e., gaining stream—moving water from the riparian zone to the channel) or lose water by outflow to groundwater (i.e., losing stream—moving water from the channel into the riparian zone). Many streams do both, gaining in some reaches and losing in other reaches. Input of cold groundwater to the bottom of pools can be a key refugia feature for anadromous fishes in summer months (Osaki 1988).

The riparian zone has been conceptualized as a zone of transmission of ground water and hillslope water to the stream channel, as well as a direct router of precipitation and snowmelt when the riparian water table rises to the ground surface. Between storms, and even during small storms with dry antecedent conditions, subsurface inputs from adjacent hillslopes are often minimal. At these times, two-way exchanges of water between the stream and the riparian aquifer (hyporheic exchange) can become important (Moore and Wondzell 2005). The hyporheic zone is an area adjacent to the channel and below the floodplain (if present) where surface water and groundwater mix. Hyporheic zones link aquatic and terrestrial systems and serve as transition areas between surface water and groundwater systems. The hyporheic zone contains species common to both surface and subsurface systems, including a diverse community of macroinvertebrates. Few hyporheic studies have focused on unconstrained headwater streams in the Pacific Northwest. Consequently, the knowledge of hyporheic hydrology draws largely upon studies of larger, unconstrained streams.

Transpiration by vegetation in the riparian zone may extract groundwater from the riparian aquifer, producing a diurnal decrease in riparian water-table level and in streamflow, followed by recovery at night. Lundquist and Cayan (2002) report that diurnal cycles are evident in many western river records and that daily variation in streamflow is often 10-20% of the daily mean flow. Harvesting in the riparian zone can have a significant influence on riparian-zone hydrology through its effect on transpiration and water-table drawdown, potentially dampening or eliminating diurnal fluctuations in discharge and increasing low-flow discharges (Bren 1997). During extended periods of low flow, sections of small streams dry up wherever stream discharge is insufficient to both maintain continuous surface flow and satisfy water losses through the bed and banks. Stream drying may occur frequently in the headmost portions of the channel network, interrupting connectivity (Moore and Wondzell 2005). Also, forestry-related changes in channel morphology can substantially influence stream-aquifer interactions. Channel incision and simplification of channel morphology during large floods can substantially lower water tables and reduce exchange flows of water between the stream and the riparian aquifer (Wondzell and Swanson 1999).

Neither the effect of forest harvesting nor the effect of riparian buffer strips on hyporheic exchange flows has been directly examined in small headwater streams (Moore and Wondzell 2005). Moore and Wondzell (2005) hypothesize, however, that because channel morphology strongly controls hyporheic exchange, it is reasonable to assume that timber operations that lead to losses in channel complexity would reduce interactions between the stream and the riparian aquifer. In contrast, they state that efforts to minimize management impacts on channels, such as retention of riparian buffer strips, would help preserve stream-aquifer interactions. The ecological implications of decreased stream-aquifer interactions are stated as being difficult to predict with current knowledge. Moore and Wondzell (2005) report that Wondzell and Swanson's research (1996) suggests that such decreased interactions could lead to reduced nutrient cycling and reductions in stream productivity.

Forest Management Impacts on Water Transfer/Exchange Processes

Forest management activities include timber falling, timber yarding, road and crossing construction and use, site-preparation activities, herbicide applications, forest thinning, etc. Forest operations on a watershed-basis can influence surface and subsurface runoff in several ways. For example, decreased interception loss increases the amount of water infiltrating the soil, leading to higher water-table levels during storms (Moore and Wondzell 2005). Limited timber falling and tree removal in riparian zones alone will reduce interception loss and evapotranspiration, but will likely have little impact on streamflow (low flows, peak flows, or annual water yield), as discussed previously. In contrast, ground-based yarding activities in riparian zones and floodplains of larger river systems can adversely impact important overflow channels used by salmonids during high winter storm discharges. Additionally, riparian areas are vulnerable to both compaction and physical disturbance during ground harvesting operations due to areas of high soil moisture and low soil strength that are common within streamside zones. These concerns, along with riparian and aquatic habitat protection, provide a basis for limiting mechanical harvesting activities within riparian zones (Dwire and others 2006).

Considerably less is known about forest management impacts associated with small headwater channels when compared to larger fish bearing watercourses. Even though streamflow is sporadic in ephemeral first order channels (typically Class III watercourses), it is capable of transporting fine sediment down to fish-bearing streams. Rashin and others (2006) found that at several study sites in Washington, delivery of sediment to unbuffered tributaries resulted in adverse impacts to fish-bearing streams that were otherwise adequately protected by riparian buffers.

Field evidence from the Caspar Creek watershed suggested that unbuffered, headwater stream channels, particularly in burned areas, contributed significantly to suspended sediment loads. Lewis and others (2001) state that sediment increases in the North Fork Caspar Creek tributaries probably could have been reduced by avoiding activities that denuded or reshaped the banks of the small headwater channels. Much of the post-harvest increases in sediment yield in the North Fork were attributed to harvest-induced storm flow volume increases (Lewis and others 2001), suggesting that the hydrologic changes can be practically and not just statistically significant (Moore and Wondzell 2005). Therefore, there is evidence that increased flows in small headwater channels, as well as disturbance of these channels, can produce increased downstream sediment transport. Further discussion of sediment delivery is provided in the California State Board of Forestry and Fire Protection's Technical Advisory Committee (TAC) Sediment Primer.

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PC 4/20/07

KEY QUESTIONS: WATER RIPARIAN EXCHANGE FUNCTION

Embedded or implied in each key question below are the following issues for the contractor's synthesis:

A. Relationship to each of California's regions;
B. Context for riparian buffer strip size: stream order, stream class, topography, hydrologic regime, climate;
C. Context for comparisons: pristine, 'optimum', legacy, or pre-harvest conditions;
D. Relationship of the quality of forest management practices being evaluated to current California forest practices; ability of BMPs to effectively mitigate identified problems;
E. Relationship of alterations to salmonid habitat quality and feeding effectiveness.

1. **How do forest management activities or disturbances in or near riparian zones/floodplains and adjacent to small headwater first and second order channels affect flow pathways and streamflow generation?**

    a) Have forest management activities in riparian zones for higher order channels with floodplains and adjacent to small headwater first and second order channels been shown to alter water transfer to stream channels, affecting near-stream and flood prone area functions (e.g., source area contributions to stormflow, bank instability, lateral and vertical channel migration, flow obstruction or diversion of flow)?
    b) Have forest management activities in riparian zones for higher order channels with floodplains and adjacent to small headwater first and second order channels been shown to result in changes in tree canopy/volume that significantly affects evapotranspiration and/or interception, with resultant changes in water yield, peak flows, low flows, etc.?
    c) Can forest management activities in riparian areas alter water yield, peak flows, or low flows sufficiently to affect channel morphology or the aquatic ecology of headwater streams?

d) Can forest management activities alter water quantity in riparian zones for higher order channels with floodplains sufficiently to affect overflow/side channels that serve as refugia for fish during floods?
e) Do forest management activities in riparian zones for higher order channels with floodplains and adjacent to small headwater first and second order channels significantly affect hyporheic exchange flows?

2. **What bearing do the findings of the reviewed articles have on <u>riparian zone buffer strip delineation</u> (area influencing water transfer/exchange function) or characteristics (cover, plant species and structure, etc.)?**
   [Note that, as opposed to the large wood and heat/microclimate functions, defining a buffer strip width for water transfer is difficult, since for any given season or year, the saturated riparian zone will vary widely]

3. **Are there regional differences in the effects of forest management activities or disturbances in or near the riparian area/zone for the water transfer riparian function? Please explain.**

PC 3/22/07

Initial List of Literature: Water Riparian Exchange Function

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PC 3/27/07

APPENDIX F: LITERATURE REVIEW SCREENING CRITERIA

1) **Literature is in Primer:** Literature the TAC used to create the Primers generally would not be included as literature the Performing Entity would review as part of the contract.

2) **Literature in Initial List of Literature to be Reviewed:** If literature which is already listed in the "Initial List of Literature to be Reviewed" in the Appendices for each Key Riparian Function need not be duplicated.

3) **Key Riparian Function:** Literature which contributes to the Key Riparian Function topics described in the Scope of Work would be considered for review under this contract. These topics address forest riparian functions for anadromous salmonid and the effects forest management has on them.

or

Scope of Work (SOW) Key Questions: Literature contributes to Key Questions developed for each r Key Riparian Function topic in Scope of Work/contract.

4) **Peer Reviewed:** Literature which is "Peer Reviewed" or meets criteria for "Non Peer Reviewed" gray literature would be considered for review under this contract.

Gray Literature inclusion criteria:

- At least three (3) of our TAC members (with multi-stakeholder perspectives) will review each of the present gray literature papers.
- Rate each paper on a 1 to 3 or 1 to 5 scale about its professional quality; its scientific contribution to understanding riparian functions
- Compare how close we come in our evaluations.

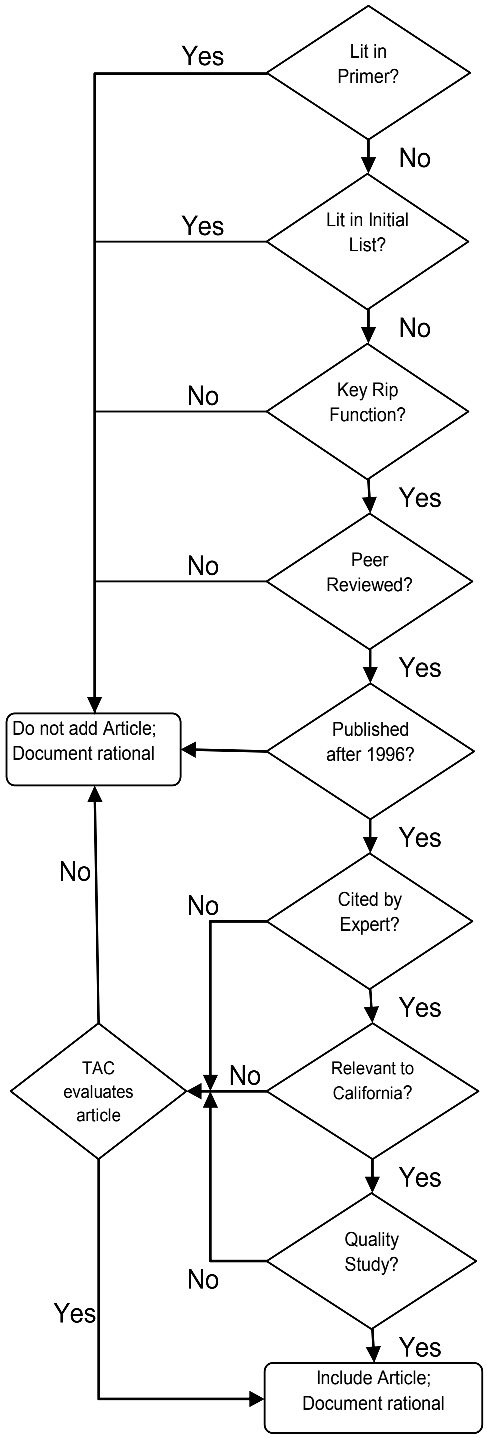
3) **Currency:** Literature which was published from 1997 to present, unless approved by Contracting Representative, would be considered for review under this contract. Literature prior to 1997 should be approved by the Contracting Representative prior to inclusion in the contract.

4) **Recommended by Expert:** Literature which was identified by experts in Top 10 list of required articles would be considered for review under this contract.

5) **Relevance to California:** Literature which applies to California or studies conducted in California would be considered for review under this contract. Relevance includes similar geomorphologic provinces or bioregions, similar topographic conditions. Geographic areas within California should be identified using Ecological Sub region terminology.

6) **Quality and Type of Study:** Literature which represents findings from large scale field experiments, models, or other descriptive studies, is data rich, theory rich, and process rich, and uses already proven study design would be considered for review under this contract.

Literature Review Decision Tree



APPPENDIX G: LITERATURE REVIEW DOCUMENTATION FORM

(Note: Responses shall be input into an Access Data base format)

(Ver: April18,2007)

Riparian Function:

Bibliographic citation: (use Microsoft Notes format or as following example:

Chen, Y.D., et al, 1998. Stream temperature simulation of forested riparian areas: 1. model application. *J. Environ. Engr.*, 124:4:316.)

Key Question(s) addressed (Site Key Questions number)

Summary of findings: A paragraph that summarizes or quotes the key findings of the study.

Location: Location of research (geographic State, County(ies), Basin/Sub-basin. Private or public land ownership)

Type of study:
Synthesis of regional literature or ecological theory
Observational study of aquatic/riparian function (esp. California)
Correlation Study
Response Study
Cause-and-effect Study
Biological Study
Forest Management Practices Study
Large scale field experiment
Manipulative Study
Fish Bearing Stream
Analytical Model

Year(s) Field Work was Done:

Study Methods: Methods used in study. Parameters measured. Methods/instruments used for measurements (e.g. canopy cover – siting tube, densitometer, solar pathfinder). Statistical analysis of the data – ANOVA, non-para, regression, etc.

Forest Management practices used for study: (e.g., managed/unmanaged, no cut/thinning, basal area/# trees, canopy retention, statistics, tools, treatments, etc…) Under what rules were the management practices being done under (e.g., Washington State Forest Practices Board; USFS – Region 5 – Six Rivers NF)? What date?)

Timeframe: Timeframe of study and monitoring; relationship to large storm events (if any). Specific dates of the study. General climate and specific weather conditions during the study period.

Study Area Characteristics Germane to Study:
Scale of Study: Stream order; acreage covered
Hydrological Characteristics: flow regime (permanent, intermittent, ephemeral); stream width, flow, etc
Stream Reach Characteristics: erosion, transport, storage; CA FPR Stream Class (I-IV); habitat type
Aquatic Biology: Aquatic species present/studied; specify which salmonid species
Riparian Zone Characteristics: Riparian species (vegetative and animal) present/studied; zone width, vegetative structure
Geomorphic/geologic features and processes: Bank/channel structure, sediment composition, stability, armoring, mass wasting, step/pool, sinuosity, LWD function, aggradation, avulsion, accretion, diversion
Forest Type and Characteristics (redwood / mixed conifer / alpine fir / oak woodland / riparian hardwood / etc.; old growth / second growth / mixed age / burned, etc.)
Geologic/ecologic province:
Climate regime/zone:

Applicability to geographic areas of California: (use Ecological Subregions of California as defined in National Hierarchical Ecological Units and provide justification/evidence of basis for Subregion selection)

Salmonid Study:

Salmonid Life Stage Function: (Select all that apply) Spawning, Incubation, Rearing, Smoltification, Feeding, Migration.

Salmonid Population Viability: (Select all that apply)Abundance, Productivity, Genetic Diversity, Spatial Distribution, Species Diversity

Reference Type: **(Circle one) Subcategories -** Literature Review, Journal Article, BookChapter, Book, Master's Thesis, PhD Dissertation, Conference Paper, Conference Proceedings, Report, Government Document, Technical Bulletin, Other (specify)_.

Peer-reviewed? Yes No Probably Don't Know

Specific Findings Pertinent to California Forest Practices within Riparian Zone:
(list)

Reviewer:

Notes: (including reviewer's perspective and applicability to riparian function and answering Key Questions)